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Proceedings of the 3rd National Ground Water Quality Symposium Held in Las Vegas, Nevada on September 15-17, 1976

National Water Well Association, Worthington, Ohio

Prepared for

Robert S Kerr Environmental Research Lab, Ada, Okla

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The Third National Ground Water Quality Symposium was held in Las Vegas, September 15-17, 1976, in conjunction with the annual convention of the National Water Well Association.

The Symposium was dedicated to the late Thomas P. Ahrens and the keynote address was given by Charles C. Johnson, Jr., Chairman of the National Drinking Water Advisory Council.

There were eight main sessions encompassing twenty-four technical papers. These were concerned with the disposal of waste on the land, the movement of pollutants in the subsurface, and artificial recharge. A special session was dedicated to ground water in the Las Vegas Valley.

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PROCEEDINGS OF
THE THIRD NATIONAL GROUND WATER QUALITY SYMPOSIUM

Cosponsored by the
U.S. ENVIRONMENTAL PROTECTION AGENCY
and the
NATIONAL WATER WELL ASSOCIATION

September 15-17, 1976

Las Vegas, Nevada

Contract No. 68-03-2396

Project Officer

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FOREWORD

The Environmental Protection Agency was established to coordinate administration of the major Federal programs designed to protect the quality of our environment.

An important part of the Agency's effort involves the search for information about environmental problems, management techniques, and new technologies through which optimum use of the Nation's land and water resources can be assured and the threat pollution poses to the welfare of the American people can be minimized.

EPA's Office of Research and Development conducts this search through a nationwide network of research facilities.

As one of these facilities, the Robert S. Kerr Environmental Research Laboratory is responsible for the management of programs to: (a) investigate the nature, transport, fate, and management of pollutants in ground water; (b) develop and demonstrate methods for treating wastewaters with soil and other natural systems; (c) develop and demonstrate pollution control technologies for irrigation return flows; (d) develop and demonstrate pollution control technologies to prevent, control, or abate pollution from the petroleum refining and petrochemical industries; and (f) develop and demonstrate technologies to manage pollution resulting from combinations of industrial wastewaters or industrial/municipal wastewaters.

This report contributes to the knowledge essential if the EPA is to meet the requirements of environmental laws that it establish and enforce pollution control standards which are reasonable, cost effective, and provide adequate protection for the American public.

William C Galegar
William C. Galegar
Director

ABSTRACT

The Third National Ground Water Quality Symposium was held in Las Vegas, September 15-17, 1976, in conjunction with the annual convention of the National Water Well Association.

The Symposium was dedicated to the late Thomas P. Ahrens, and the keynote address was given by Charles C. Johnson, Jr., Chairman of the National Drinking Water Advisory Council.

There were eight main sessions emcompassing 24 technical papers. These were concerned with the disposal of waste on the land, the movement of pollutants in the subsurface, and artificial recharge. A special session was dedicated to ground water in the Las Vegas Valley.

The Transactions of this Symposium are submitted in fulfillment of Contract No. 68-03-2396 by the National Water Well Association under the sponsorship of the U.S. Environmental Protection Agency.

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PROCEEDINGS OF THE THIRD NATIONAL GROUND WATER QUALITY SYMPOSIUM

September 15-17, 1976

Las Vegas, Nevada

WASTE DISPOSAL — A MOST PERVASIVE PROBLEM[a]

by Charles C. Johnson, Jr.[b]

I am most pleased to be here today to speak before The Third National Ground Water Quality Symposium—in part because it is my first opportunity to speak before such a large and distinguished audience since becoming a member of the National Drinking Water Advisory Council, and in part because of my belief that we must constantly be aware of potential insults to all of the mediums—air, land, and water—that comprise our environment. Of these, our water resources, more perhaps than any other, illustrate the interaction of all parts of the environment and particularly the recycling process that characterizes every resource of the ecosystem. Everything that man himself injects into the biosphere—chemical, biological or physical—can ultimately find its way into the earth's water. And these contaminants must be removed, by nature or by man, before that water is again potable.

But mostly, I am pleased because I believe that once again in this decade the United States is on the verge of another environmental awakening, and this time it is directly focused on man's insults to his environment as they impact on the health and well-being of people. This is evident by the sustained attention and debate by the public on the proposed toxic substances' legislation; the broad citizen acceptance of the principle of nondegradation of our air resources; the firm stand evolving with respect to continued and forceful implementation of the water pollution control program; and the several court decisions that support the rights of individuals to be protected from potential as well as proven harmful effects attributable to their environments. Symposia such as this that bring together experts of proven scientific competence and experience to exercise their mature judgment in the light of all existing data, will provide the fuel that is necessary to sustain this continuing and expanding concern for problems associated with the environment while contributing to the solutions of environmental problems we so desperately need.

Close analysis may reveal that America's biggest problem, and indeed maybe its biggest enemy, is its ever-growing and yet unmanaged production and disposal of waste products. This apprehension is true whether our concerns embrace social, political, or economic progress;

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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ending of inflation and unemployment; increasing international trade and producing a balance of payments; expanding agricultural production or producing the goods desired for modern living; or creating the energy independence so important to our survival as a Nation. The production of unwanted waste products is part and parcel to every action and reaction that accompanies proposed, accepted, and implemented solutions to all of the societal problems that befall our communities and our Nation. Our approach to these solutions affect us as a people and affect our place in the world in which we live.

A discussion of the tremendous volumes of waste produced in this country tends to boggle the mind. U.S. industries treat about 5,000 billion gallons per year of wastewater before discharging it into the environment. Of this volume, about 1,700 billion gallons are pumped to oxidation ponds or lagoons for treatment or as a step in the treatment process. Another 5,375 billion gallons per year of municipal wastewater is discharged to the environment, with an estimated 730 billion gallons of this amount discharged to the land. In the United States, municipal sludge production amounts to four million dry tons per year. Industrial sludge production is believed to be many times this amount. Some estimates place annual municipal solid waste production at more than 170 million tons per year and growing. To these figures must be added the millions of tons of gaseous wastes that are produced annually; the untold hundreds of millions of tons of mining tailings disposed of each year; and the tremendous volumes (more than 24 million barrels per day) or oil field brines produced each day.

As a result of all the actions and interactions that accompany and follow the production of these wastes, we continue to create the kind of environment that fouls our air, pollutes our streams, desecrates our lands and endangers our people. In the process we create enmity between persons, problems between communities, disputes between states, and division between countries. We set people against industry, industry against government; and place the government, seemingly, against both industry and the people. These reactions are manifest in part by the many court suits associated with efforts to regulate waste disposal practices and to locate waste disposal sites at the local, regional, state, and national levels. No one wants another's waste, and neither do we want our own.

In our zeal to correct the problems we have created for ourselves, we have almost legislated the Nation into a no-win situation. We have a national air pollution control law that says you can't put the waste in the air; a water pollution control law that says the Nation's goal is zero discharge; and a safe drinking water law that says you cannot contaminate the domestic water supply. The only relatively unregulated medium available for disposal of our unwanted waste products is the land.

There are those who argue, "Why allow the discharge of any waste? Why not create the conditions under which all industries—public and private—recycle or reuse all waste products?" You and I know that the simple answer is that, sooner or later, there is an irreducible residual which, even under theoretical conditions, must be disposed of or stored in a safe repository. In actual practice geography, economics, waste characteristics, and technology preclude total recycling and reuse of many waste products. We have seen that the last repository not totally regulated for the disposal of these wastes is the land. History has demonstrated that waste products tend to gravitate for disposal purposes to the point of least regulation.

This then brings me to the point I wish to make today: Our continued failure to recognize, appreciate, deal with or accept the fact that the environment in which we live is interconnected and interrelated in every aspect of its being—its air, its water, and its land—can undercut all of the good intentions manifest in the legislation and programs that have been implemented to restore the integrity of the environment.

The evidence suggests that we are blindly engaged in activities that could despoil two important and necessary resources—our land and the ground-water aquifers that exist underneath it. In our eagerness to make progress in dealing with the problems of the environment—and we are making progress—we seem to be pursuing a course, where land and waste products are concerned, of "out of sight, out of mind." We are permitting—and even promoting in some circumstances—under the guise of environmental conservation, or as accepted waste disposal practice, or under the practice of zero discharge of liquid wastes, the deposition of treated and untreated sewage and other hazardous wastes onto and into our lands and over our ground-water aquifers with little, and in some instances, no restraints.

I am told that in this country at least one-half of the population depends upon ground water as a source of drinking water. Of the total population, 29 percent use ground water delivered by community systems, and another 19 percent have their own domestic wells. These Americans would like to think—to believe—that because of the recent concern for the environment, and the volume of legislation that has been enacted to support this concern, their ground-water supplies are protected against the insults of man-made pollution. Unfortunately, this is not the case. While both the Water Pollution Control Act and the Safe Drinking Water Act serve much needed purposes, neither provides the kind of authority required to really protect our ground-water supplies from possible degradation. Whether they should or not suggests the basis for an interesting discussion.

The basic and primary and essentially sole interest of the Water Pollution Control Act Amendments of 1972 is directed to control the pollution of surface waters. In fact, sections of the Act that require consideration of new and alternative waste treatment and disposal methods are used to support and promote the use of land application of sewage and other liquid wastes and their residuals as a method of achieving zero discharge. When a land application methodology is used, no discharge permit is required unless runoff is collected for discharge from a point source to a surface stream. In the absence of a permit, there is no legal responsibility or authority in the Federal program to require discontinuance of the practice, with one exception—the Administrator may act under his emergency powers to protect the public health. Few states and localities have standards and regulations to deal adequately with these practices.

What are the implications of this situation with respect to our ground-water supplies? A recent publication of the EPA states, "The ground-water quality and level will be affected by most land application systems. Exceptions would be many overland flow, underdrained, and pumped withdrawal systems. Wastewater constituents that are not used by the plants, degraded by microorganisms, or fixed in the soil may leach to ground water" (U.S.E.P.A., 1975). In a literature review in 1974, EPA concluded, "Although the published coverage on the various sources of ground-water pollution is uneven, it appears that the number of locations where pollution occurs is very large; however, the area affected is believed to be quite small" (U.S. E.P.A., 1974).

Gerba, *et al.* (1975) of the Department of Virology and Epidemiology at Baylor College of Medicine, within the year published an article entitled "Viruses in Water: The Problem and Some Solutions." In the article, the authors warn that "while land disposal of domestic wastewater has been practiced on a large scale in Europe for several decades, little is known about the fate of viruses during and after application to the soil." They continue, "It has been feared that without dilution in receiving waters, a greater threat will exist to ground water if sewage is discharged directly onto the soil in large quantities. It is also felt that the deep well injection of wastes may pose similar problems if not controlled." Discussion and examples included in the article are supportive of this concern. The authors observed that between 1946 and 1960, 61 percent of all waterborne outbreaks of disease in the United States were caused by contaminated ground water (Gerba *et al.*, 1975).

Schiffman recently reported on a survey of hazardous waste disposal in the State of Maryland and concluded, "It now appears that all or nearly all existing discharges of industrial wastes to the ground and underground waters are causing some environmental damage. The basic question is the degree of impact" (Schiffman, 1975). The report noted that in the State, approximately 2.4 million tons of industrial waste are generated annually from the chemical industry alone. It further noted that 75 percent of 530 million gallons of liquid waste are disposed of to the ground at the place of generation. Underground water is the supply source for over 40 percent of the people in Maryland and is the source of stream base flow. What is the danger to our underground aquifers if Maryland's experience is translatable to the Nation's total industrial capacity?

Unfortunately, our concern cannot stop with the treatment and disposal of the liquid fraction of wastewater flow. By 1985, municipalities alone are expected to produce some 260 million wet tons of sludge annually. This material cannot legally be placed in our streams or dumped in the ocean, and its incineration is discouarged in most localities. As a result, it must be placed on the land. By simple deduction, sludge has to be as dirty as effluent is clean. It contains all the elements which have been removed to protect our surface streams and our oceans. Why then is its uncontained and uncontrolled placement on the land such an improvement in the protection of the environment?

The Safe Drinking Water Act was enacted in part to provide the protection to our ground-water supplies that is not obtainable under the water pollution control legislation. A reading of the House report easily allows one to draw this conclusion. The report noted that "underground injection of contaminants is clearly an increasing problem. Municipalities are increasingly engaging in underground injection of sewage, sludge, and other wastes. Industries are injecting chemicals, byproducts and wastes. Energy production companies are using injection techniques to increase production and to dispose of unwanted brines brought to the surface during production. Even government agencies, including the military, are getting rid of difficult-to-manage waste problems by underground disposal methods. Part C is intended to deal with all of the foregoing situations insofar as they endanger underground drinking water sources" (Safe Drinking Water Act, 1974). By way of identification, Part C in the Act is entitled "Protection of Underground Sources of Drinking Water."

Without a doubt this is a very broad and laudable mandate. I for one would like to believe that this indeed was the intention of Congress when it enacted the legislation. I was further encouraged by the words in the House report that defined the term "injection." They say, "The definition of underground injection is intended to be broad enough to cover any contaminant which may be put below ground level and which flows or moves, whether the contaminant is in semi-solid, liquid, sludge or any other form. . . This definition is not limited to the injection for disposal purposes."

What could be clearer? In the broadest possible context, in the words of the Committee and subsequently the Congress, the primary concern of this portion of the Safe Drinking Water Act is the protection from contamination of the Nation's ground-water resources that are or may be used for domestic water supplies. In the drafting of the legislation, however, a problem arises. As the lawyers interpret the operative words in the Act, the words "well" and "injection" are regarded as a unit and as such limit the scope of authority inherent in the Act to protect the ground water. Thus, subsequent regulations can only regulate the practices whereby subsurface emplacement of wastes occurs through "well injection."

By definition, pits, ponds, and lagoons—which many believe to be the major contributors to ground-water pollution—and the practice of land application of wastes under the impetus of zero discharge are not included in the proposed underground injection control program. At the Federal level, these practices can only be regulated by the emergency powers granted the Administrator under the Act. If it is true that the legal interpretation of the legislation precludes provision for the protection that is required for underground-water sources, it is most important that the public and the Congress be made aware of this problem and that corrective legislation be enacted to clarify the situation.

We must constantly remind ourselves that contamination of ground-water aquifers can occur easily and rapidly. Once they are contaminated it may take many years and could cost millions of dollars to purge the aquifer of the contaminant. Unless we are willing to treat the water from these underground sources as we do that from our surface streams, we must exert every effort to prevent their becoming contaminated.

Lest we become too complacent in the progress that is being made through our legislative and regulatory process and programs, it should be recognized that even under the practices permitted by the proposed underground injection regulations, insofar as protection of the public's health is concerned, there is some risk, supported by considerable ignorance. My reading tells me that there is much more unknown than is known about the significance of the quality of waste discharged to our underground aquifers and the ultimate fate of those wastes once they have been placed in the ground. Some studies have been conducted on the travel of some pollutants, under some conditions. Few studies have fully addressed, and none have resolved, our current concerns for water reuse, ground-water recharge, discharge of highly concentrated effluents and sludges, or pretreatment requirements as related to the inherent relationship of these practices to long-term exposure of people to the trace elements ultimately associated with these wastes. We are told that the significance of the long-term ingestion of low levels of pollutants of water is difficult to ascertain. We read that their effects are insidious and are often screened by other factors with other effects. We know that most of these elements are not good for us; we just don't know when, and how bad, they are for us.

Okun (1976), writing in the *American Journal of Public Health,* referenced a London study that indicated that persons drinking water from a safe underground source experienced lower cancer mortalities than persons in another population group in the same city provided with treated water from a polluted source. In the same article, he noted that "studies made in Holland showed the municipalities receiving their drinking water from polluted rivers had a higher cancer death rate than those taking their water from purer underground sources." These references suggest that once a water supply is polluted, even treatment of the water does not assure the level of protection we desire for the public compared to the use of a protected and unpolluted source. Why then should we even consider a disposal methodology for situations that have the potential for contamination of naturally safe ground water?

Many years ago, a decision was made to use our streams, our rivers, our oceans, and the air over our lands for the disposal of the waste products of our progress. That decision was made in ignorance and supported by convenience. Waste volumes were small and disposal costs were low. Today we are paying the price for our lack of knowledge and absence of forethought. Notwithstanding the untold illnesses and deaths that some relate to these past activities, the many billions of dollars allocated from tax dollars and by industries to reclaim our streams and our air is testimony to the folly of our past actions. Doesn't this history teach us something? Shouldn't we avoid repeating our past mistakes? Will we take the steps that are necessary to preserve our land for the production of crops and our underground-water resources for domestic water-supply needs?

Earlier on, I made the statement that, in addressing our requirements for cleaning up our environment, we had almost legislated ourselves into a no-win situation—don't discharge wastes to the air, don't discharge wastes to the water. Now it appears that I am suggesting a zero discharge to the land. Lil Abner, Al Capp's famous comic strip character, probably would say, "As any fool can plainly see, it's got to go some place." And it must. Under the circumstances, we people have some hard choices to make. These choices concern beneficial use, risk analysis and priorities associated with our knowledge of and appreciation for the interrelationship of air, land, and water in our total environment. We must decide what waste will be tolerated in the environment and in what amounts. We must determine under what circumstances it is better for us to deposit our waste products on the land, in the air or in the water. We must recognize that in the disposal of wastes, convenience, expediency, and low costs may be shortcuts to environmental deterioration and endangerment of human lives. We must accept the fact that knowledge and understanding of the forces of the environment influence the sum total of our actions as a Nation and are essential to the protection of our people and the preservation of the earth.

I would not want to leave the impression that no waste materials can or should be placed on, in, or under the ground surface. Given the proper hydrogeological conditions and using appropriately designed facilities, there are situations when selected wastes can be disposed of into the ground without appreciably modifying the quality of the potable ground water. Disposal of brine from petroleum exploration and development is one example. Another situation may require the use of a nonpotable water to enhance the quality of an unusable supply or to protect the quality of a potable supply, as in the case of agricultural production or salt-water intrusions, when the required results warrant the implied risk. Further, a decision could be made that an underground aquifer's most beneficial use is for the disposal of waste products. In each of these situations, the choice is deliberate. It is made with full knowledge of its subsequent effects, either actual or potential. When we proceed in this manner, we bring order to our actions and our environmental control reguations can then be designed to have both direction and purpose. To do less is to risk repeating the history of pollution that is so indelibly associated with our surface waters and our air.

The results of your deliberations during the course of this Symposium can make a substantial contribution to the Nation's quest for knowledge and in the shaping of decisions which will guide future activity in this subject area. I wish you well as you undertake the earnest debate that will shape your sessions, as you attempt to resolve honest differences of opinion that are sure to arise, and as you labor to reach the common concensus which undergirds our democratic process.

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Thomas P. Ahrens — Father of Water Well Technology[a]

by Michael D. Campbell[b]

On May 1, 1976, Tom Ahrens died of lung cancer at the age of 69 after a long illness. In my opinion, no worker in the field of ground water contributed more to the practical aspects of water well design, construction and ground-water development than Tom Ahrens. His absence is keenly felt by all who knew him—his family and friends, as well as his professional associates both in the United States and overseas, where most of his efforts were directed after his retirement from the U.S. Bureau of Reclamation in 1972.

Tom Ahrens was born on February 14, 1907, in Chicago. His father died when Tom was 4 years old, leaving his mother to raise Tom, two older brothers and an older sister. In his early teens, the family moved to White Plains, New York where his mother was a registered nurse, subsequently becoming Director of Nursing at the hospital in White Plains. Tom's childhood and teens were happy years in a closeknit family where all the children worked after school to help support the family. Tom put himself through Georgetown University in Washington, D.C., where he met and married Arletta Phillips in the early 30's. He went on to Harvard for a Master's degree in Geology, while Arletta, being multilingual, worked for a number of embassies in Washington, D.C.

It was at Harvard that Tom's interest in ground water first developed, and it continued to grow later on during courses at Northwestern and other universities that offered engineering and ground-water-related topics. But in those days, the field of ground water had not yet received the attention it was to enjoy in later years. In the early 30's, only a handful of professionals were familiar with the rudimentary concepts of ground-water flow and even fewer knew of the applications to water well design and operation. As late as the mid 30's, C. F. Tolman at Stanford had not yet completed his monumental text *Ground Water*, C. V. Theis was still working on the significance and nature of the cone of depression, and Morris Muskat, J. King Hubbert and C. E. Jacob had yet to make their individual contributions to the field. Armed with the works of pioneers like Darcy, Dupuit, Meinzer, and others, Tom was one of the first to apply the principles of ground water to the real world, wherever water supplies were needed. It was at this stage of Tom's life that his wife, Arletta, played such a significant role in supporting Tom's attitude toward his professional goal . . . to help people, wherever they were, get water where it was urgently needed for human existence and development. They were prepared to go wherever the need was greatest, which at times turned out to be in the most far-off and desolate parts of the earth.

In 1936 Tom joined the U.S. Department of Agriculture as a water planning analyst and was located first in Washington, D.C., and then Mesilla, New Mexico and Denver, Colorado. With the onset of World War II, he was called back to Washington to work for the Military Geology Branch of the U.S. Geological Survey. In 1945, Tom went to work for the U.S. Bureau of Reclamation where, as he once expressed it, "he could deal more directly with the critical field problems of water well construction and operation than with any other outfit." During this same period, Tom helped to organize the National Water Well Association and his indelible impact can still be felt. He was the first professional geologist to stress the point that technical people must learn to serve as technical translators of the principles of ground-water geology for the water well contractors, if they are to be expected to do their jobs effectively. His wife, Arletta, continued her enthusiastic support, being a dynamo of enthusiasm in NWWA activities. Arletta died in 1957, leaving Tom and an adopted son, Pat. In the years before her death, during their foreign assignments, she and Tom were avid collectors of South American art, acquiring numerous articles of Indian dress and ceremonial objects. They donated most of the pieces to the Denver Museum of Natural History, where they are still on display today.

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[a] Memorial address presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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A few years later, Tom met and married Helen Seep, an occupational therapist with degrees from Santa Fe State and Columbia University. Having also recently lost a special spouse, Helen found unusual compatibility and comfort in the relationship. Tom had found a soul-mate. After my first extended visit with Tom and Helen in Denver, it was very apparent that their honeymoon had never really ended.

By the time Tom retired from the Bureau in 1972, he and Helen were seasoned world travelers who had represented the Bureau and the United States with a dignity and honor that did justice to our highest diplomatic ideals. In their own country as well, they were gracious hosts to a constant stream of associates and friends from all over the world. Theirs was an open-door policy of true hospitality. Guests were always welcome in their Denver home—drilling contractors, geologists and engineers alike—as those of you who knew them can attest.

In March of 1975, during one of his many post-Bureau overseas consulting assignments, Tom contracted pneumonia in Pakistan, which began a long period of decaying health. He refused to give up his work commitments, however, and a few months later, in May, he took on a series of lectures at the Indian Reservation at Window Rock, Arizona where a serious attack brought his active contribution to an end. Cancer, surgery, radiation treatment and chemotherapy occupied the painful 12 months to follow. In spite of all, he maintained his wonderful sense of humor, his courage and his remarkable patience to the very end.

Although Tom produced numerous reports and publications, most of his energies were directed toward the technical and personal development of many of his younger associates, one of whom was myself. His philosophy was that of a practical and far-sighted man of action. One of Tom's beliefs was that an error-free individual is an unproductive individual, and that without mistakes real progress cannot be achieved, either professionally or personally. Tom felt that many scientists, especially earth scientists, are so restricted in their thinking that many of their careers are spent avoiding mistakes rather than accepting their inevitability and profiting from them. He believed that all individuals must be innovative to be productive and that the fear of failure must not paralyze the will to achieve.

Tom gave his time, his thoughts and his energies unselfishly. He was the guiding light for many of the projects and programs in the ground-water field, especially within the U.S. Bureau of Reclamation and the NWWA. It was Tom Ahrens who began the dialogue between the well driller and the technical man. It was he who urged, years ago, an engineering approach to the design of water wells, especially in cementing and grouting and casing selection. And it was Tom who quietly pressed for effective water well construction standards and efficient rural water systems.

Tom was one of the early ground-water salesmen, but where most good salesmen tend to exert uncomfortable pressure, he was low-key and gentle, but at the same time would put forth a strong case for his beliefs. He preferred to be part of a crowd until something important to him would stir him to speak up. The confidence, experience and well-considered position that would come forth were always rewarding to the listener.

Tom was a Registered Professional Engineer in the State of Colorado, a member of the American Geophysical Union, National Water Well Association (Honorary Life Member), American Water Resources Association, Geological Society of America, Rocky Mountain Society of Geologists, American Society of Professional Engineers, and chief consultant to the NWWA Research Facility since 1972. He published ground-water reports on Somalia, East Africa, Chile, India, and gave numerous presentations to societies, commissions and associations. He was awarded a special service citation by the U.S. Department of the Interior and numerous other merit awards by scientific and industrial groups.

Tom's contribution to his field will be felt for many years to come. He maintained a natural comradeship with most everyone. He placed integrity, personal as well as scientific, above all else. The field of ground water could not afford to lose him and those who knew him well will long miss him. He was, indeed, the father of water well technology, and it is fitting and proper that this Third National Ground Water Quality Symposium is dedicated in the memory of Thomas P. Ahrens to record our collective appreciation of his professional contribution to the field of ground water and our admiration of his personal contribution to the many people who were guided toward a better understanding of our science and of ourselves.

Wastewaters in the Vadose Zone of Arid Regions: Hydrologic Interactions[a]

by John F. Mann, Jr.[b]

ABSTRACT

In truly arid regions there is essentially no direct penetration of rainfall. Recharge to ground water is not only infrequent, but extremely localized, occurring only where surface runoff has been channelized or ponded. Over those vast desert areas covered by sparse xerophytic vegetation, the scant rainfall has little or no chance of becoming ground-water recharge. Such water is quickly dissipated by capillarity-assisted evaporation, or through rapid evapotranspiration by short-lived annuals. Where perennial xerophytes cover the ground surface, the extensive shallow root systems quickly utilize all of the rainfall stored in the soil. Beneath the infrequently moistened soil zone is the lower part of the vadose zone, extending to water tables which are usually at depths of tens to hundreds of feet. Almost always these vadose zones have moisture contents well below field capacity.

Regardless of the cause, these dry vadose zones are capable of holding additional water, at least up to field capacity. And no water-carried pollutants can reach the water table from the ground surface until a pre-wetted path has been formed for the entire vertical distance. A practical use of this water-holding capacity can be made in the design of wastewater tailings ponds, with predictable safety and with great economic benefit. However, only in predictable geologic conditions, and in limited amounts can the use of this water-holding capacity be recommended. On the other hand, to make no use whatsoever of these great natural dry sponges would be an economic waste.

INTRODUCTION

The term "vadose water" was originally used by Posepny (1894) to include all of the water in the zone of aeration. Later, Meinzer (1939) divided the zone of aeration into three parts and proposed a series of terms of Greek origin to apply to the contained waters. For all of the water in the zone of aeration he proposed the term "kremastic water." He proposed "rhizic water" for the water in the soil zone, "anastatic water" for the water in the capillary fringe, and "argic water" for the water in the intermediate zone. Unlike "phreatophyte" which has become a commonly used term among hydrologists, Meinzer's adaptations from the Greek for waters in the zone of aeration have never caught on, and few ground-water geologists normally use them. As regards research activities, the intermediate portion of the zone of aeration must be considered a true orphan (Bean *et al.*, 1967). The surficial materials to the depth of root penetration (soil, hydrologically speaking) have been the subject of intensive research by soil scientists. Some agriculture-oriented hydrologists such as Blaney (1933) have systematically calculated deep penetration of rainfall primarily from a soil-moisture approach. The water which is not disposed of by evapotranspiration or stored in the soil zone is *presumed* to reach the water table. The soils engineer, to whom soil may be anything which is not hard rock, is concerned mainly with strength characteristics, but does collect samples of the intermediate portion of the vadose zone and makes moisture determinations. Actually, a great deal of soil moisture information has been collected by soils engineers in arid regions, but remains in their files because the information was not relevant to anticipated problems. Thus we find the vadose zone of arid regions an inadvertent victim of non-emphasis, with considerable data existing, but with few organized attempts to study and understand its hydrologic aspects. From laboratory studies, Palmquist and Johnson (1962) have demonstrated the mechanism of movement of water in vadose zones, and suggested the possible use of arid vadose zones for nuclear waste disposal. Winograd (1974) raised the possibility of storing high-level radioactive wastes within containers in the thick unsaturated zones of arid regions.

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THE VADOSE ZONE IN ARID REGIONS

It is generally recognized that there is no direct recharge by rainfall through the soil zones of arid regions. Xerophytic vegetation, by definition, grows in an environment of sparse soil moisture. Following a rainstorm, such vegetation quickly removes the water stored in the soil until the plant's capacity to remove water is equalled by the soil's tenacity for holding water. In this environment of great competition for soil moisture, the perennial xerophytes are typically widely spaced and shallow rooted. In addition, short-lived annuals quickly remove moisture from the shallow soils in places where they can compete successfully against capillarity-nourished evaporation and the far-flung root systems of the perennial xerophytes. Many xerophytes are capable of utilizing more water than reaches them as direct rainfall. Examples can be seen in the Mojave Desert of California where the most abundant xerophyte is creosote bush (*Larrea divaricata*). Individual plants growing close to paved roads are commonly two to three times as large as those growing on the normal alluvial surfaces. The paved roads typically have shallow drainage ditches on both sides into which is directed the rain which falls on the impervious pavement. Those creosote bushes whose roots are able to reach the augmented rainfall in the soil below the ditches grow with unusual vigor. As yet, no research has been done on the question of whether there is penetration of rainfall beyond the rooting depth in these ditches. Quantitatively, such recharge to ground water, if any, would be small.

Studies of recharge in arid regions, particularly by the United States Geological Survey in Arizona and Nevada show that the main recharge to the alluvial basins occurs from the beds of flowing streams within a short distance after those streams have left the mountains. Winograd (1974) has provided us with the following summary:

> "Hydrologic, geomorphic, and pedologic evidence suggest that little or no recharge (namely, infiltration of precipitation to the water table) occurs beneath interfluves (inter-stream areas) in the arid and semiarid portions of the Southwest under present climatic conditions."

In truly arid regions the above generalization seems to hold, but as pointed out by the writer previously (Mann, 1957) there is considerable penetration of rainfall (especially in very wet years) in semiarid regions. As average annual rainfall increases, a point should be reached where direct penetration of rainfall may be expected to start. Many other variables such as the permeability of the soil, angle of slope, and nature of vegetation would be involved. In a study of the Mojave River basins in California, Chun and Weber (1967) used an average annual rainfall of 8 inches as the upper limit of the zone of no direct rainfall penetration. The same figure of 8 inches was used by Malmberg (1967) in studies in Pahrump Valley.

In those areas where there has been no downward passage of infiltrating rainfall for unknown thousands of years—perhaps since the ending of the last Pluvial period—moisture levels have dropped far below field capacity. This condition has been confirmed in numerous borings supervised by the writer in the Mojave Desert area. Where air rotary rigs are used, these dry vadose zones are characterized by lost circulation problems and by dust blowing out of the hole. The mechanism by which the vadose zone drops from the field capacity condition it is assumed to have had at the time of the last infiltrating water to its present low-moisture condition is poorly understood. Evaporation by air movement accompanying changes of barometric pressure is probably a factor. The production of hydrated weathering products is another. Minor, but perhaps reversible, changes may be due to vapor transfer.

The depth of the dry vadose zone is highly variable. Winograd (1974) speaks of depths of 100 to 300 meters in local areas. Regardless of the depth, there are in places thick zones with moisture levels far below field capacity. And in considering the applications of liquids to interfluve soils we may liken these vadose zones to huge dry sponges. Before liquids can reach the water table they must travel a path which has been pre-wetted to field capacity. The volumetric pellicular film capacity which can be suspended above the water table could be viewed as a permanent storage reservoir for certain liquid wastes.

MOVEMENT OF WATER IN DRY VADOSE ZONES

Some interesting field tests bearing on the mechanism of movement of water in dry vadose zones were conducted during 1970 at the site of a coal-fired generating plant in southern Nevada. One of the wastes to be disposed of was coal ash. Prior to the start of operation of the plant, a question was raised as to whether rain, falling upon ash

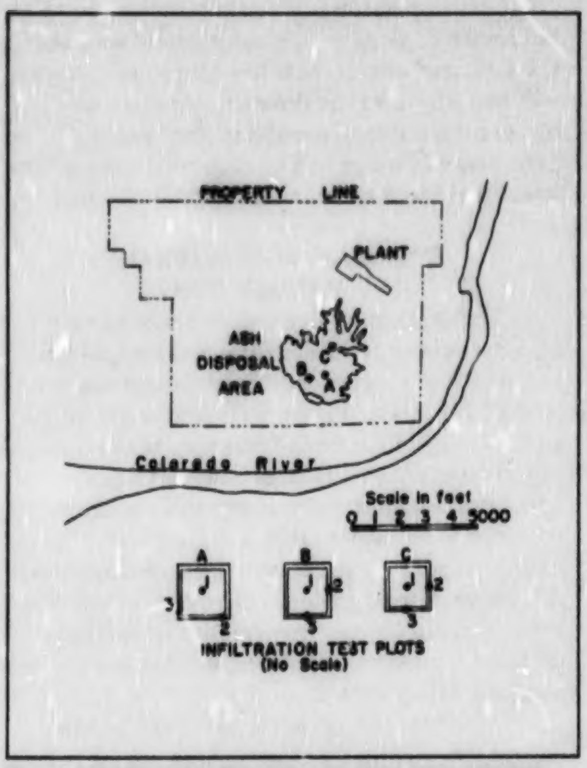

Fig. 1. Locations of infiltration test plots near coal-fired generating plant (after Glenn A. Brown).

which had been spread on the ground surface, would cause pollution of the underlying ground waters, which in turn were thought to be tributary to the Colorado River. It was known from laboratory tests that ash leachate had a total dissolved solids content in the range of 3000 to more than 5000 parts per million. Prior to disposal of the ash, three infiltration test plots were constructed (Figure 1). Each plot was about 10 feet by 10 feet (3 m X 3 m) and about 18 inches (0.5 m) high. At each plot three holes were drilled and equipped with 1 5/8-inch (4.13 cm) shelby tubes. The central holes were 40 feet deep, and plywood sheets were placed around the holes to prevent disturbance to the natural soil by the drilling rig. At each plot 2 holes were drilled in the berms to depths of 25 feet and also equipped with 1 5/8-inch (4.13 cm) shelby tubes.

The ash disposal area is in a dissected alluvial surface extending easterly from the Newberry Mountains. Several generations of terrace deposits overlie an extensive sequence of silty and clayey lake beds.

Prior to the flooding of the test plots, the natural soil moisture was measured in each hole by means of a Nuclear-Chicago Model P19 moisture probe. Such neutron-scattering moisture meters have been used by soil scientists for many years (Nixon and Lawless, 1960) and have been confirmed as acceptable substitutes for the older, more laborious methods of determining soil moisture (de Boodt, Moerman and de Boever, 1969). Both the neutron probe measurements and standard laboratory tests on core samples indicated that the moisture conditions in the vadose zone, above the water table at a depth of about 85 feet (26 m), were considerably below field capacity.

More specific results are shown for Hole A-1 (Figure 2). The materials encountered in the entire 40 feet of hole consisted of fine to coarse sand with 10 to 20 percent gravel. A 6-inch (15 cm) plug of concrete was poured around the tube to prevent direct entry of water down the outside of the tube. Water was introduced to the plot once, to a depth of 5.25 inches (13.3 cm). Its downward progress was then monitored by means of the neutron probe. On the first day, the top 4 feet (1.2 m) were nearly saturated. By the second day, some of the water from the top 4 feet (1.2 m) had moved down to 10 feet (3 m). There appears to be a translocation of gravity water from the shallow zones and a downward progression of a pellicular front in the deeper zones. The rate of reduction of

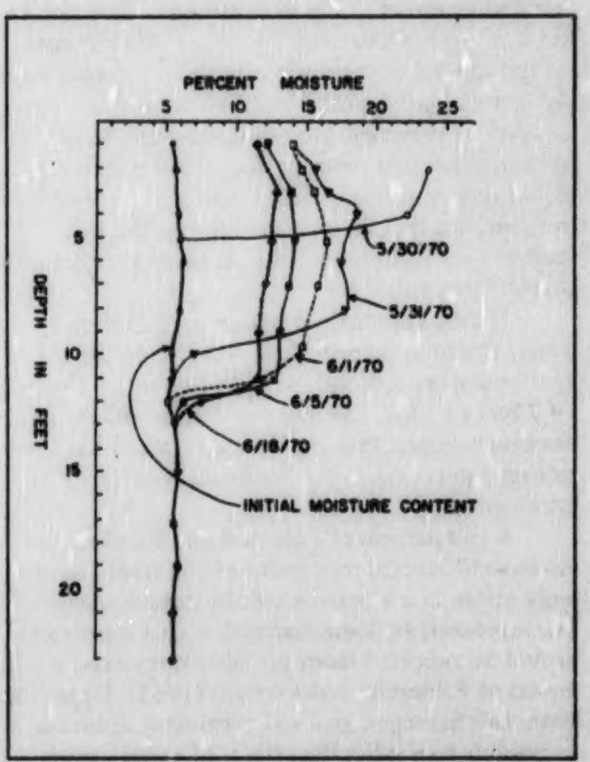

Fig. 2. Progress of wetting front in Hole A-1 (after Beene Engineering).

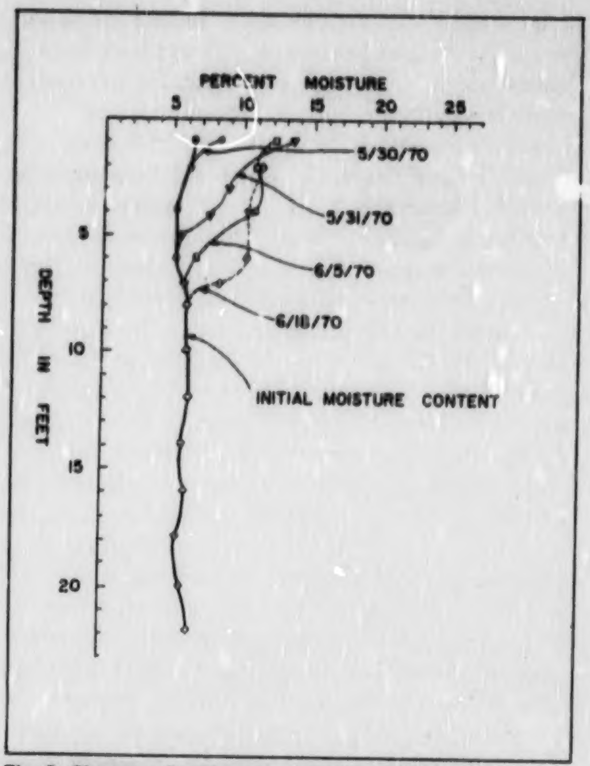

Fig. 3. Changes of moisture content in Hole A-2 (after Buena Engineering).

moisture content in the shallow zones slows with time as there is downward drainage. Almost none of the applied water reached to depths greater than 12 feet (3.7 m), below which the natural moisture conditions prevailed. From this diagram, in a relatively homogeneous geologic situation, one could suggest that the field capacity was 10 or 11 percent, but that factors operating in the arid vadose zone had reduced the moisture content to about 5 percent.

The curves for Hole A-2 (Figure 3) show a vastly different picture. This shows some lateral movement beneath the berm, but only to a depth of 7 feet (2.1 m). The low percentages of moisture suggest that capillarity, rather than gravity saturation, is the dominant wetting mechanism.

A comparison of Figures 2 and 3 indicate a downward vertical movement of the water, with only minor lateral movement. In this relatively homogeneous geologic material, such a movement would be expected from the laboratory experiments of Palmquist and Johnson (1962). Departing from this homogeneous and permeable situation, it is possible to predict the effects of certain types of geologic heterogeneities. If there are nearly horizontal low-permeability layers, perching is to be expected, along with greater lateral spread of the saturated zones. Such low-permeability layers will not only slow the downward rate of movement but there is a strong possibility that a greater percentage of water will be required to bring the lower-permeability materials up to field capacity.

DISPOSAL OF WASTES OF DRY VADOSE ZONES

The capacity of dry vadose zones to tie up liquid wastes as pellicular films and wedges can be utilized to great economic advantage and with predictable safety. Figure 4 illustrates the simplified concept of controlled leakage. At the prospective site of a tailings pond for liquid wastes which are not chemically acceptable for direct recharge to the water table, a drilling and coring program would have to be undertaken to determine natural geologic and moisture conditions. From this information the amount of leakage permitted could be calculated, of course with an adequate safety factor.

Industrial tailings ponds are often of huge dimensions—hundreds of acres in extent. The costs of low permeability liners are enormous. Within this economic framework, there is justification for

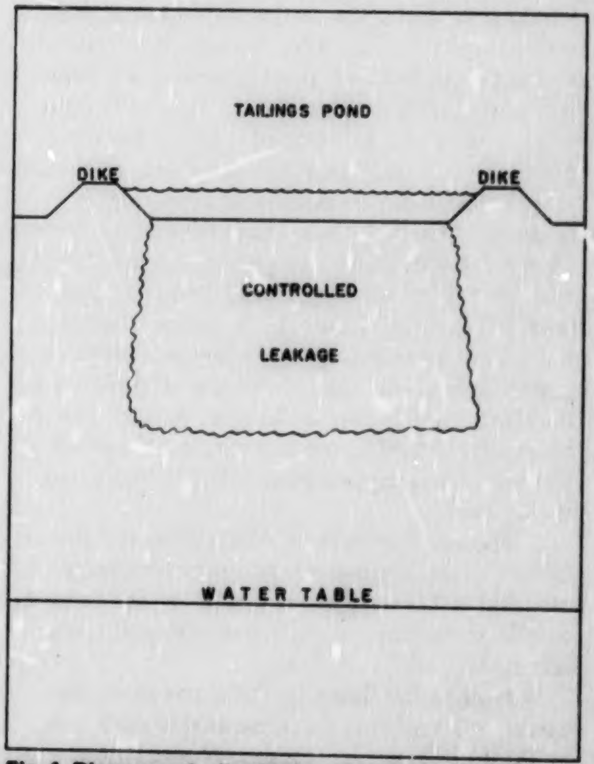

Fig. 4. Diagrammatic distribution of moisture beneath a tailings pond under a controlled leakage approach.

considerable testing and experimentation based on sound geologic and hydrologic principles.

In the Mojave Desert of California, studies were made for the installation of a 129-acre tailings pond for the evaporation of highly saline wastes from a large chemical plant. Local clays were available to construct a compacted low permeability liner. Laboratory tests indicated that a permeability of 0.002 feet (0.06 cm) per year could be achieved. The clay layer (compacted in two lifts and 0.83 feet [25 cm] thick) would be covered by a permeable protective blanket to avoid desiccation of the clay layer if the ponds were emptied. Pre-flooding with fresh water was planned to provide a reservoir of fresh water within the protective blanket which could saturate the clay and negate capillary forces within the clay. The saline tailings water would then follow the fresh water through the clay. Under a "zero leakage" concept there is no provision for pellicular storage in the underlying dry vadose zone; on the contrary, the useful life of the pond is considered to end when the tailings water reaches the bottom of the base (clay) layer. Pond life as a function of total pond head, using a design permeability of 0.002 feet (0.06 cm) per year and an effective porosity of 0.33 is shown in Figure 5 for different thicknesses of the impervious clay layer. A pond built with a clay layer 0.83 feet (25 cm) thick and a head of 10 feet (3 m) is expected to have a useful life of about 12 years, at which time a new tailings pond would have to be completed and ready to receive tailings water. However, by adopting a controlled leakage approach, the useful life of the pond could be increased. As this pond receives solid wastes (slurry) as well as liquids, it will progressively lose its capacity for seasonal storage of liquids. Its useful life could be extended further if the dikes could be raised to permit a higher theoretical head. Actually, during the deposition of the water-carried solids, there are laid down nearly horizontal clay layers, so that the effective head operating on the base layer may be less than would be indicated by Figure 5.

Under the design criteria used, Figure 6 shows the relationships among total pond head and base (clay) thickness as it relates to the rate at which water will move through the base (clay) layer. Using such data, along with information as to moisture content of the underlying vadose zone, it is possible to predict the downward movement of the wetting front and to stop it at a safe distance above the water table.

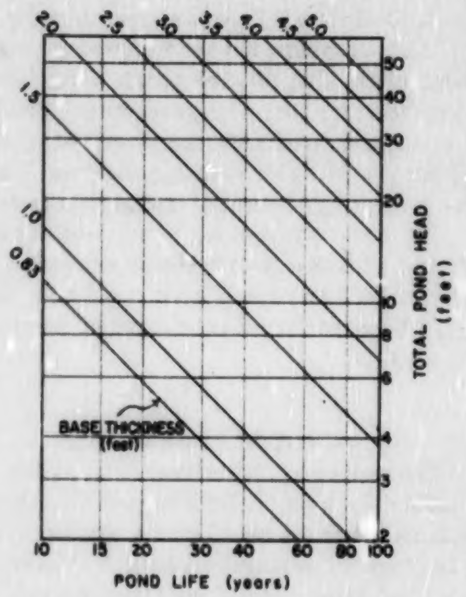

Fig. 5. Relationships among pond liner thickness, total pond head, and pond life under a zero leakage concept (after Moore and Taber).

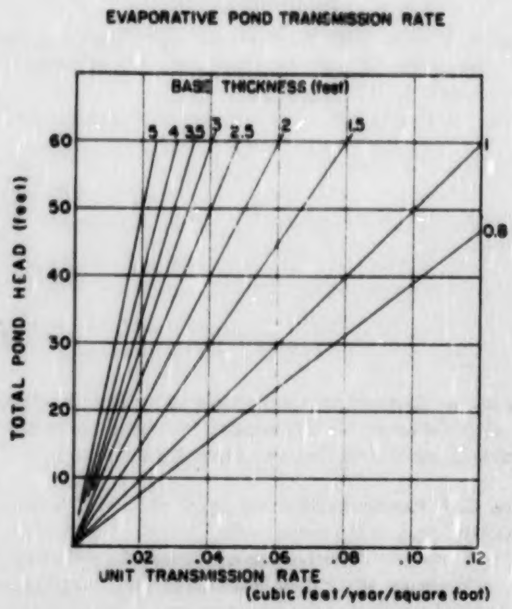

Fig. 6. Rate of leakage through clay layers of various thicknesses at various total pond heads (after Moore and Taber).

MONITORING

The key to acceptance of a controlled leakage approach is monitoring. Classical monitoring of the water table downgradient from the tailings pond may be unacceptable if such monitoring results in the detection of a pollution problem only after it has reached unmanageable proportions. The facts demonstrating direct downward vertical movement suggest monitoring wells within the pond itself, or such monitoring of perched lateral flow as would be indicated by the geologic conditions. With such wells and a neutron probe, the downward movement of the wetting front could be followed, and discharge of the effluent could be stopped where desired.

ACKNOWLEDGEMENTS

For making available basic data used in this paper, and for many helpful suggestions, the writer would like to thank Mr. Glenn A. Brown, Consulting Geologist, Los Angeles; Mr. R. F. Moore, of Moore and Taber, Consulting Engineers and Geologists, Anaheim, California; and Buena Engineering, Ventura, California. However, neither their cooperation nor helpfulness should be construed as agreement with all of the writer's conclusions, or any of them.

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DISCUSSION

The following questions were answered by John F. Mann, Jr., after delivering his talk entitled "Wastewaters in the Vadose Zone of Arid Regions: Hydrologic Interactions."

Q. by G. F. Hendricks. *Please redefine the term "field capacity." How is it determined?*
A. Field capacity describes the condition of a soil after gravity drainage. It is a term most often used in agricultural circles, and similar to specific retention.

Q. by Ron Barto. *Do you think that the low field capacity is due to evaporation?*
A. Evaporation is a possibility. On a longer-term basis, hydration as feldspars are weathered to clays, is another.

Q. *In the case of the Nevada Ash Pit, at about what level of natural rainfall would the vadose zone reach field capacity?*
A. There are some suggestions that average annual rainfall as little as 8 inches per year would result in some deep penetration of rainfall on coarse permeable soils. Tighter soils and hot season rainfall would probably place this value higher.

Q. by Jack E. Sceva. *How do you evaluate the potential for the development of perched ground-water zones and the lateral movement of leakage away from the site?*
A. By thorough coring of the site and determination of permeabilities. Low permeability layers may be expected to cause perching and lateral spread.

Q. by K. Childs. *Shouldn't the integrity of the suggested leaking pond be verified by a ground water and soil zone monitor network, or don't you feel it is necessary?*
A. I would suggest both neutron probe monitoring within the pond and water-table monitoring downgradient.

Q. by Dan Kimball. *Are there any ground-water observation wells located around the ash pond at the Mojave plant and, if so, have they shown any changes in ground-water quality?*
A. No changes in ground-water quality in the monitor wells have been related to the disposal of ash.

Q. by Bill Bellis. *With all the core samples – what is the mineralogy?*
A. There has been visual logging of the cores and cuttings but no detailed mineralogical study.

Q. by Ed Meiser. *What is the danger or likelihood of localized, pre-wetted paths developing as routes of direct communication between the source of wastewater application and the water table?*
A. In usual alluvial environments, this should be no problem. Concern should be expressed in areas of cracking due to compaction or tectonics. A thorough subsurface investigation must be made.

Q. by Jan Turk. *In your "no leakage" example, fresh water was used to saturate the clay lining before the highly saline waters were allowed to pass through the clay. Have you noticed any shrinkage of the clay when the brine arrives (i.e. osmotic desiccation)?*
A. The brine is discharged as a slurry, which causes the bottom to be covered by deposited clays. Furthermore, the clay is covered by a permeable layer, pre-wetted with fresh water. Under these conditions, desiccation is unlikely.

Q. by Jack Robertson. *Does the clay lining of ponds produce any ion filtration effects?*
A. No liquid has shown in the underdrains, so it hasn't been possible to check this.

Q. by D. G. Boyer. *What is your type of clay? Bentonite or non-expanding clay? What is the concentration and make-up of your saline fluid?*
A. The clay is not highly expansive. The saline fluid is a high sodium brine with 50,000 to 100,000 parts per million total dissolved solids.

Q. by C. W. Fetter. *Homo sapiens have existed, so far, through a number of climatic changes. During the late Pleistocene, the "arid" southwest had heavy precipitation. Does waste disposal in the arid zone present potential problems to future generations if the climate shifts to a humid one? Long-forgotten contaminants might become mobile.*
A. The possibilities of a climatic change to wetter (or drier) over a span of thousands (rather than hundreds) of years are a consideration. Covering with a low permeability blanket has been suggested as a means of preventing future leaching by infiltrating rainfall or flood flows.

Wastewaters in the Vadose Zone of Arid Regions: Geochemical Interactions[a]

by Donald D. Runnells[b]

ABSTRACT

Because of increasingly stringent laws governing discharge of fluid wastes to surface waters, the alternative of discharge to the subsurface has become attractive. The physical-chemical processes that prevail in the subsurface are not well understood, but they are clearly not identical to processes of purification in surface waters. For example, in the subsurface the process of oxidation may be of little value in significantly reducing the concentration of discharged contaminants; in contrast, oxidation plays an important role in purifying surface waters. Eleven physical-chemical processes can be identified as having potential value for purifying wastes discharged to the subsurface, as follow: dilution, buffering of pH, precipitation by reaction, hydrolysis, oxidation or reduction, filtration, volatilization, biological assimilation, radioactive decay, membrane filtration, and sorption.

Discharge to the vadose zone may be a safe means of disposal of wastes in arid regions. But it is necessary to carefully test the suitability of a particular site for a particular waste. Processes of purification in the vadose zone can be incorporated into a workable plan of discharge if adequate studies and safeguards are employed. Regulations governing subsurface discharge should take into account the physical-chemical processes that may act to purify the waste fluids. In one set of experiments, a soil from Sulfur Springs, New Mexico was capable of removing large quantities of dissolved molybdenum and copper from a synthetic mill water, and the soil was able to quantitatively retain the copper during subsequent leaching by fresh and metal-free mill waters. Such studies permit rational plans of discharge to be developed.

INTRODUCTION

Until fairly recent times we have generally assumed that ground water was well protected from contamination. And in many areas this seemed to be a fairly realistic view. Now, however, we know that the apparent purity of ground water was the result of our failure to *look* for pollution. With increasing pressures of population, and with incessant prodding from new regulations, water scientists have begun to fully recognize the widespread occurrences of contamination of ground water. Table 1 shows chemical components that have been demonstrated as being present in documented cases of contamination of ground water throughout the world (summarized from several sources, including Furiman and Barton, 1971; Scalf and others, 1973; Miller and others, 1974; Cole, 1974; van der Leeden and others, 1975). Each of these cases demonstrates that the protective capacity of subsurface materials is finite, and that this protective barrier can be overwhelmed by excessive discharge of virtually any contaminant.

The means by which contaminants may enter ground water are many and varied. Figures 1 and 2 illustrate some of the more obvious sources and pathways of contamination of ground water. The details of the hydrology of movement of

Table 1. Chemical Components Present in Documented Cases of Contamination of Ground Water Throughout the World

chromium
cadmium
zinc
lead
fluoride
iron
barium
manganese
nickel
silver
molybdenum
boron
uranium
mercury
aluminum
lithium

cyanide
copper
selenium
chloride (and hydrochloric acid)
sulfate (and sulfuric acid)
nitrate
detergents
radium and other radioactive wastes
phenols
alcohol
gasoline
leachate from landfills
pesticides
herbicides
solvents

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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Fig. 1. Summary of some obvious pathways by which contaminants may enter ground water.

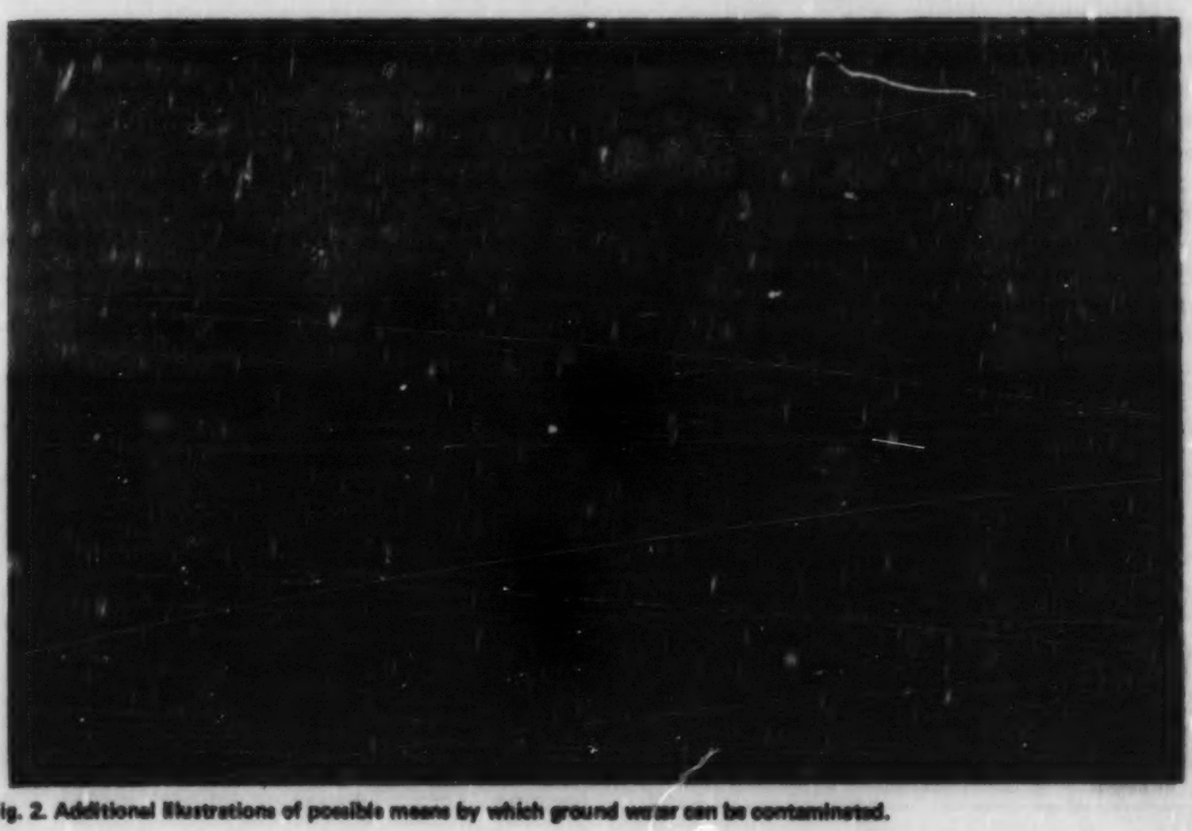

Fig. 2. Additional illustrations of possible means by which ground water can be contaminated.

contaminants in the subsurface is treated in other papers in this Symposium and will not be discussed here.

Despite the fact that we are aware of hundreds of documented cases of ground-water contamination, the sobering fact is that there certainly exist thousands of similar cases of which we remain blissfully unaware. Miller and others (1974) made the important observation that most instances of ground-water contamination become known because of obvious leaks or spills at the surface, or because of complaints from subsequent users. Contamination of ground water is generally *not* discovered because of monitoring or routine chemical analysis. Many States in the past have not required analyses of water supplies for many toxic substances, such as barium, selenium, and silver, despite the listing of such substances in the 1962 Drinking Water Standards of the U.S. Public Health Service.

POSSIBLE PROCESSES OF PURIFICATION

Introduction

Having now emphasized the fact that ground water can surely become contaminated, let us look at some of the processes which offer protection to ground water, even in the face of significant discharges of potential contaminants to the subsurface. In Table 2 are listed eleven physical-chemical processes that may, under favorable circumstances, afford significant protection to the chemical purity of ground water. These eleven processes are listed together for the sake of completeness, but all are surely not equally effective.

The possible role and importance of each of these processes is discussed briefly in the following sections. Special emphasis is placed on their role in purifying wastes that might be discharged to the vadose zone in arid regions.

Table 2. Eleven Physical-Chemical Processes That May Operate in the Subsurface to Purify Fluid Wastes

1. Dilution.
2. Buffering of pH.
3. Precipitation by reaction of wastes with indigenous waters or solids.
4. Precipitation due to hydrolysis.
5. Removal due to oxidation or reduction.
6. Mechanical filtration.
7. Volatilization and loss as a gas.
8. Biological assimilation or degradation.
9. Radioactive decay.
10. Membrane filtration.
11. Sorption.

Dilution

The first process listed in Table 2, dilution, is well understood as it occurs in surface waters. However, we cannot rely on dilution to be effective over the short term in the vadose zone, especially in an arid region. And in any location, we must expect dilution of wastes *below* the water table to take place far more slowly than in surface streams. Indeed, the one outstanding characteristic of contamination of ground water is the slow rate at which it is diluted and dispersed by the natural flow system. Fryberger (1972) has presented convincing data and arguments to show how expensive and difficult it may be to reclaim a contaminated aquifer, even considering natural attenuation due to flow and dilution.

Buffering of pH

Natural ground waters in arid regions generally exhibit pH values between about 6 and 9. The upper limit is established in nature by the reaction between carbon dioxide gas and either limestone or caliche in the soil. The lower limit is an educated guess at the pH that might be found in shallow ground water from non-reactive rocks, such as quartz sandstone, or in ground water issuing from a mountain meadow. Trost (1974) found a range in pH from 6.0 to 9.0 in 1392 samples of ground water from southern Arizona.

The pH is a critical factor in many reactions involving contaminants, including processes that affect the stability of solid minerals and precipitates in the subsurface. If a discharged fluid has a pH outside of the range of 6 to 9, it is possible that drastic chemical reactions may occur, including the dissolution of such natural solids as calcium carbonate, iron oxyhydroxide, and manganese oxyhydroxide. Such reactions can exacerbate problems of contamination of ground water.

Precipitation by Reaction

This process could be quite important in purification of discharged fluids. A perusal of any handbook of chemistry will show that *in theory* it is possible to precipitate virtually any dissolved contaminant if the appropriate precipitating agent is present. In nature, however, the necessary precipitants do not commonly exist in sufficient quantities or favorable locations to cause removal of toxic substances to safe levels. For example, it is theoretically possible to reduce the concentration of dissolved sulfate in a discharge by mixing it with a natural barium-rich ground water. However, barium-rich ground waters are not common, so this particular theoretical control will generally not operate in nature. In natural ground waters and in the enclosing solids, only certain species are normally present and available for reaction in meaningful concentrations. These species include dissolved calcium, magnesium, sodium, potassium, bicarbonate, sulfate, chloride, and silica. In the surrounding porous solids we find the additional abundant components of iron, aluminum, and possibly carbonate and phosphate. Dangerous or undesirable contaminants that might be reduced to safe concentrations by reaction with these species include the following: arsenic (precipitation by iron, aluminum, and calcium), barium (by sulfate and carbonate), cadmium (by sulfate and carbonate), copper (by carbonate), fluoride (by calcium and phosphate), cyanide (by many metals, including iron), iron (by silica), lead (by carbonate, sulfate, and phosphate), mercury (reduced form precipitated by carbonate and chloride), molybdenum (by iron and aluminum at proper pH), silver (by chloride), zinc (by carbonate and silica), and radium (by sulfate and carbonate). Evidence that such precipitation reactions do occur and may control concentrations in ground water has been given by many researchers. For example, Dutt and others (1968) found that the concentrations of iron and zinc in the ground waters of the Tucson Basin are apparently controlled by dissolved silica and the pH. However, despite the numerous possibilities for purification offered by precipitation reactions, a great many examples of contamination of ground water exist involving species that might be expected to precipitate; such examples demonstrate the failure of precipitation to protect ground water in many cases.

Hydrolysis and Precipitation

This type of reaction occurs when a dissolved contaminant reacts with water, with the release of either a hydrogen ion or a hydroxyl ion. The best known example involves the hydrolysis of dissolved ferric ion with precipitation of ferric hydroxide. On the acid side, molybdenum is thought to hydrolyze to bimolybdate ion and precipitate with compounds of iron and aluminum under acid conditions (LeGendre and Runnells, 1975). In an aerated environment, such as the vadose zone in an arid region, the concentration of dissolved iron can be held to very low values by hydrolysis and precipitation of ferric oxyhydroxide (Hem, 1970). However, in oxygen-deficient zones, such as commonly exist in the phreatic zone, high concentrations of iron can occur in the ferrous (Fe^{2+}) form. High rates of infiltration or flooding by waters into the vadose zone, such as might occur beneath a disposal lagoon, may produce anaerobic conditions and lead to solubilization of some metals.

In addition to ferric (Fe^{3+}) iron, several other toxic metals form highly insoluble hydroxides in the range of pH of 6 to 9. These include copper (above a pH of about 6.5), chromium (the +3 form only, above a pH of about 6), nickel (above a pH of 9), and zinc (above pH 7.5). For example, Dutt and others (1968) found that the concentration of copper in ground water of the Tucson Basin seems to be controlled by either cuprous (Cu^{+}) or cupric (Cu^{2+}) hydroxide.

The apparent importance of hydrolysis in both the vadose and phreatic zones is the result of the simple requirement that water be present, in contrast to other precipitation reactions that demand the presence of specific precipitating agents.

Precipitation Due to Oxidation or Reduction

Relatively insoluble oxides of certain contaminants are known for copper, iron, manganese, mercury, and nickel. Aerobic conditions favorable for the development of these oxides will normally be found in the vadose zone and possibly in the upper parts of the phreatic zone. Deep in the phreatic zone or in swampy or flooded vadose environments, anaerobic conditions may prevail and lead to the mobilization of some species. Dutt and others (1968) in their work in the Tucson Basin found that rapid infiltration of domestic sewage caused a loss of oxidizing conditions in the vadose zone and led to increased movement of copper, zinc, manganese, nickel, and lead through the soil.

In some instances reducing conditions are most favorable for removing possible contaminants from water. One example is that of chromium, which is highly soluble and mobile in the oxidized state (+6, chromate or dichromate ion), but quite insoluble as the solid oxide or hydroxide of the reduced form (+3). Natural reducing conditions can also theoretically cause the formation of such native elements as arsenic, copper, mercury, selenium, silver, and lead, each of which is quite insoluble. We do find significant quantities of native copper, mercury, and silver in mineral deposits, and native selenium appears to be a stable phase in some soils (Goering and others, 1968). In reducing environments bacteria can convert dissolved sulfate to sulfide and dissolved nitrate to ammonia or nitrogen gas. Reaction of the sulfide with certain metals will produce highly insoluble precipitates, such as the sulfides of arsenic, cadmium, copper, iron, lead, mercury, molybdenum, nickel, silver, and zinc.

In terms of usefulness and reliability for purification, the processes of oxidation and reduction can be of significant value if the necessary environmental conditions are maintained and managed. Considerable reliance can probably be placed on the process of oxidation for purification of wastes in the vadose zone. Processes of reduction may be more risky because some undesirable species can be mobilized, such as iron (Apgar and Langmuir, 1971) and manganese. One well-known benefit of the process of reduction is the denitrification of dissolved nitrate from sewage effluent, with significant removal of nitrogen possible under a carefully programmed mode of disposal (Bouwer, 1974).

Mechanical Filtration

The sixth process listed in Table 2, mechanical filtration, will generally be of little help in removing dissolved contaminants. Exceptions to this might occur following the formation of precipitates by means of one of the processes mentioned earlier. Such precipitates would be filtered out mechanically as the water moves through the porous medium. Iron is the species most likely to exhibit this behavior; it is well established that much of the total iron in aerated waters is present as particulate ferric hydroxide (Hem, 1970). Manganese probably behaves in a similar fashion. If these possible particulates sorb or include other dissolved species as they form, the filtration would remove these other species as well. Of course, most particulates originally present in the discharge would be filtered out quickly during downward movement through the vadose zone.

Volatilization and Loss as a Gas

Some inorganic species can be effectively removed from discharged waters by volatilization. Perhaps the best known example is that of the bacterial reduction of dissolved sulfate to hydrogen sulfide gas, with loss of the hydrogen sulfide to the atmosphere (Kellogg and others, 1972). Mercury in solution can also be volatilized in anaerobic environments (Lagerwerff, 1972) or by reaction with dissolved humic acids (Alberts and others, 1974). Several organic compounds of arsenic are volatile, and escape of arsenic as a gas has been demonstrated for both aerobic and anaerobic soils (Woolson and others, 1971). Based on its similarity to sulfur, we might also expect selenium to be subject to volatilization (Lakin, 1973). And of course, the microbial reduction of nitrate to ammonia and nitrogen gas is well documented (Bouwer, 1974), although the failure of this mechanism to protect ground water is demonstrated by numerous examples of pollution of ground water by nitrate (Minear and Patterson, 1973).

Volatilization as a means of purification is poorly understood, and we probably run considerable risk in relying upon it to any great extent. The exceptions to this would be the conversion of dissolved sulfate and nitrate to gases; these processes are known to be effective if managed properly.

Biological Degradation

Biological degradation or assimilation, the eighth process, is very important in removal of organic and biologic contaminants. Many organic substances would be removed or attenuated by biologic activity in the subsurface, especially in the oxidizing environment of the normal vadose zone. The biologic involvement of sulfate and nitrate have already been mentioned. In addition, arsenic, cyanide, mercury, and selenium are likely candidates for biologic fixation or volatilization. Molybdenum is strongly assimilated and concentrated by plants that are nitrogen-fixers (Johnson, 1966). And in their study in the Tucson area, Dutt and others (1968) found that grass assimilated and removed significant quantities of metals from infiltrating sewage effluent.

Biological processes could be of great value in managing discharges to the subsurface, but at present we know so little of the principles involved that each case must be studied and evaluated on its own.

Radioactive Decay

This mechanism is of value in the management of radioactive wastes by means of storage in the subsurface. Winograd (1974) has recently discussed the attractiveness of storage of high-level radioactive wastes in the vadose zone of arid environments. It seems clear that storage in this environment is possible with a high degree of safety for periods of time from thousands to hundreds of thousands of years, during which time the wastes would lose much of their activity through processes of decay. There is a substantial history of disposal of radioactive wastes to the subsurface near nuclear facilities in the United States, and in humid regions serious problems of contamination of ground water can be expected. It is the lack of recharge to move contaminants to the ground water that makes disposal to the vadose zone in arid regions so attractive.

Membrane Filtration

The tenth possible process of purification, membrane filtration, is a topic that is greatly in vogue at the present time in the literature of the geochemistry of subsurface waters. A summary of the phenomenon, given by Hanshaw (1972), points out that some observed data on the pressure and salinity of formational waters cannot be explained by simple gravitational flow of water or by dissolution of minerals. In such cases, the anomalous data may be related to osmotic pressures and filtration of salts across beds of shale. Pressure differentials of several hundred atmospheres can theoretically be generated across a membrane if fresh water exists on one side and a saturated brine on the other (Hanshaw, 1972).

In terms of purification of wastes in the subsurface, it is theoretically possible that a discharged wastewater could be diluted by osmotic transfer of water across a clay-rich aquitard. Conversely, one can speculate on the possibility of forcing waste fluids through a clay liner on the bottom of a disposal pond, anticipating some removal of dissolved salts and passage of a relatively more pure fluid. However, at this stage in development of the technology of disposal, careful pilot studies would be required to test the efficacy of such a scheme.

Sorption

Finally, and importantly, ground water may be protected by various processes of sorption. Clays, metallic oxides and hydroxides, and organic matter can all be good substrates for sorption of various dissolved species. With the exception of fractured shale or siltstone, consolidated bedrock will generally not be very effective as a sorbent. Virtually every ionic species and many non-electrolytes will be sorbed and removed to some extent as ground water moves through an aquifer; of the ions, only chloride, and to a lesser extent sulfate and nitrate, seem to pass through soils and alluvium without significant sorption.

The troubling aspect of sorption as a means of purifying waste waters in the vadose zone is that the process can be highly specific in its action, both as to the dissolved substance and the solid substrate. Molybdenum offers an interesting example. Dissolved molybdenum is sorbed strongly by most soils that exhibit an acidic reaction with water, but molybdenum will move freely through alkaline soils (Katz and Runnells, 1974). Similarly, fluoride is sorbed much more extensively by acidic soils than by alkaline soils (Bower and Thatcher, 1967). Perhaps the most revealing study of the effect of pH on sorption of ions is that by Griffin and others (1976), in which they determined the extent of removal by clay minerals of a suite of heavy metals from synthetic solutions and natural leachates from sanitary landfills. They found a very great variation in the extent of sorption, depending on the metal involved and the pH of the solution.

It is clear that the processes of sorption depend on the type of contaminant and on the physical and chemical properties of both solution and porous medium. Under favorable circumstances sorption can be wonderfully effective in purifying waters, as evidenced by the extensive use of ion-exchange media in water treatment. One should keep in mind, however, that when a contaminant ion is sorbed, some other change must also take place to compensate for loss of the charged species from solution. In ion-exchange reactions, a different ion is released by the solid to the solution. The ions released to water from a previously uncontaminated soil or alluvium will almost certainly be less harmful than most of the contaminants of concern here.

We can place considerable reliance on the processes of sorption to help us protect ground water, but each case will be different and tests must be run to assure an adequate degree of sorption and retention.

Summary

Eleven processes have been discussed, each of which is capable, under favorable circumstances, of contributing to the purification of liquid wastes discharged to the subsurface. However, hundreds of documented examples of ground-water contamination show that in specific instances the defensive mechanisms have been overwhelmed by discharged contaminants. It is reasonable to make an attempt to use the purifying properties of the subsurface as part of a plan of discharge, especially in the vadose zone in arid regions. However, such a plan of discharge must incorporate careful studies of the capacity and efficiency of the vadose zone for removing contaminants, and of possible later remobilization of the toxic substances.

USING NATURAL PROCESSES OF PURIFICATION IN A PLAN OF DISCHARGE

Introduction

Because of the potential value for purification and long-term storage of wastes in the vadose zone of arid regions, it seems reasonable to include the use of this zone in plans of discharge. In fact, Winograd (1974) presents a detailed analysis of such a plan for storage of radioactive wastes. As water scientists, we can envision similar useful applications in the discharge of other wastes. Bouwer (1974) has given an example of proper management of disposal of sewage effluent to the vadose zone in Arizona in order to maximize removal of dissolved nitrate. At the time of this writing, the New Mexico Water Quality Control Commission is considering, for possible adoption, new regulations for protection of ground water. These proposed regulations, developed chiefly by the New Mexico Environmental Improvement Agency, would permit a discharger to make use of a fraction of the purifying capacity of the vadose zone as part of his plan of discharge to the subsurface. The concept of using the vadose zone in this manner was suggested by the present writer, as a consultant to the New Mexico Environmental Improvement Agency. It is gratifying that most parties concerned, from industry on the one hand to environmental groups on the other, seem to agree that the vadose zone in arid regions can offer a safe repository for contaminants, provided that very careful study and monitoring are incorporated into the plan of discharge.

Potential Capacity of the Vadose Zone for Purification

The capacity of the vadose zone can be large for removal and storage of contaminants. As an example of this large theoretical capacity, let us consider a hypothetical discharge of dissolved cadmium from a metal-plating plant. The assumptions involved in this example are summarized in Table 3. The concentration of 3.7 mg/l dissolved cadmium is that actually reported by Lieber and others (1964) for the well-known example of ground-water contamination on Long Island. It would be dangerous to allow a discharger to use the total capacity of the vadose zone for sorption of contaminants from his wastewater, so for purposes of this example we will assume that State regulations allow him to use 25 percent of the estimated total sorptive capacity of the unconsolidated mantle beneath his discharge pond. Based on the data listed in Table 3, the discharger could store approximately 380,000 pounds (172,400 kg) of cadmium in the vadose zone. This is a *lot* of cadmium. At a concentration of 3.7 mg/l, this amount corresponds to about 38,000 acre-feet (4.7 × 10^{10} liters) of wastewater that could be discharged from the one-acre pond.

Table 3. Assumptions Used in Hypothetical Example of the Capacity for Sorptive Removal of Dissolved Cadmium from Discharge to the Vadose Zone in an Arid Region

Assumptions:

1. Unconsolidated mantle with 10 weight percent clay.
2. Cation exchange capacity of 50 milliequivalents per 100 grams (0.22 pounds) of clay.
3. Density of 2 grams/cc (124.9 lbs/cubic foot).
4. Thickness of mantle = 100 feet (30.5 meters).
5. Area of disposal = 1 acre (0.405 hectare).
6. Efficiency of sorption is 100 percent.
7. Only 25 percent of the total sorptive capacity is utilized.

Of course, the preceding simplistic example ignores competition or enhancement of sorption by other ions, other chemical reactions, possible channelized flow of the wastewater through the vadose zone, and a great many other complicating factors. The example does point out, however, that a significant degree of purification of wastewater and storage of dissolved contaminants is possible under the conditions that exist in an arid region.

Dangers and Necessity for Study

Having emphasized the great potential value of the vadose zone for discharge, we must temper our enthusiasm by recognizing that the vadose zone has a finite capacity for purification, and that there is always a danger of leakage and contamination of ground water. Referring back to Table 1, we can see that it is possible to overwhelm the protective mechanisms in the subsurface with any dissolved contaminant. This knowledge warns us of the necessity for thorough and competent studies of the hydrologic and chemical characteristics of a proposed site *before* discharge begins. A proposed plan of discharge must also allow for a wide margin of error in the measurements required, including such parameters as heterogeneity in the hydraulic properties of the earth materials at the site. Finally, an adequate scheme for monitoring must be included, together with contingency plans in the event that the discharged contaminants do not behave as expected.

Necessary Measurements and Testing

Our knowledge of the specific physical-chemical processes that obtain in the subsurface is primitive. We cannot predict from theory what will happen when a particular contaminant in a particular fluid matrix is discharged into a specific type of soil. We do have some general knowledge of the principles involved, as discussed in an earlier section of this paper, but we remain ignorant of many of the specifics. Now that the Safe Drinking Water Act is a reality, we can expect to see a rapid increase in research and knowledge concerning the movement of contaminants through earth materials. In fact, it would be possible to list several new publications that deal specifically with this subject. By far the most complete and satisfying such report known to this writer is that by Weir and others (1975), prepared for the Electric Power Research Institute. The purpose of the work reported by these workers was to determine if toxic ions would be leached from ash and sludge discharged into ponds by electric utilities, and to determine if such ions would move through underlying soils into ground water. Another excellent study, with greater emphasis on the hydrologic factors, is that by Papadopulos and Winograd (1974).

In order to have confidence in the probable efficacy and safety of a plan of discharge to the vadose zone, a great many factors must be considered. For example, one must have a fairly complete knowledge of the hydrogeologic conditions and homogeneity of the earth materials at the proposed site of disposal. In order to obtain such information it will certainly be necessary to conduct fairly extensive field studies, possibly including drilling, to determine the rate and paths of movement of the fluid discharge. An example of one aspect of the work that may be necessary can be found in the study of seepage of effluent from septic tanks, published by Bouma and others (1972). Hajek (1969) has given a good summary of some of the technical aspects of the tests and calculations that should be done to understand and predict the chemical interactions of wastewaters with soils. And finally, a potential discharger must look to the future, realizing that "purification" of wastewaters by the vadose zone really represents *storage*, either long-term or short-term. For some chemical species, such as phosphate, zinc, and copper, the storage may be permanent because these ions can become fixed in the structure of minerals and not be available for later release by percolating solutions (Ellis and Knezek, 1972). In other cases, contaminants that have been discharged and stored in the vadose zone may be readily available for leaching and remobilization by the next soaking rainfall, or by a subsequent change in the chemical composition of the discharged fluid. Tests must therefore be run, not only to determine the extent of removal of contaminants from the discharge, but also to determine the possibility of later remobilization and flushing to ground water. With regard to storage, Winograd (1974) has discussed the selection of a site for disposal to best avoid exhumation by the normal processes of erosion or remobilization by climatic change over periods of hundreds of thousands of years.

Discharge of any kind may pose a long-term threat to the environment, and at present, our knowledge of interactions in the vadose zone does not permit us to make unequivocal predictions from theoretical principles. Much work must be empirical, and teams of scientists and engineers must be involved in devising the plan of discharge. Perhaps most importantly, each type of discharge and potential site must be treated individually, on a case-by-case basis. No one can claim that the job of insuring non-destructive disposal of wastes to the vadose zone will be easy, but neither can they claim that the job is impossible.

An Example of Geochemical Testing

As part of the task of developing possible regulations to govern discharges to the subsurface in New Mexico, this writer was asked to present an example of how the geochemical portion of the testing might be conducted. The work was not intended to be complete, but was designed instead to illustrate the minimum information that would be appropriate. A very brief summary is presented here.

The soil chosen was from Sulfur Springs, New Mexico. It was dark brown in color, contained rootlets, and represented a composite of the upper six inches of the profile. A 1:2 soil:water slurry yielded a potentiometric pH of 6.4. The soil was air-dried and sieved to (−) 2 mm to remove the coarse, non-reactive fragments. A small column of soil [approximately 23.2 cc (1.42 cubic inches)] was prepared in a chromatography tube. The test solution was gravity-fed into the column at various rates, from about 0.5 to 1.6 ml/min (corresponding to a velocity of about 4 to 12 feet/day). Saturated conditions were maintained, together with a relatively high rate of flow, because of limitations of time. An actual case would require slower rates of flow and an unsaturated column. Aliquots of the effluent were collected in volumes of about 9.5 ml (0.0025 gallons) by means of an automatic fraction collector. The porosity of the soil column was estimated in two ways: (1) addition of water to saturation and reweighing in the column and (2) measurement of volume of water withdrawn from a graduated cylinder by suction into the soil column. Density was determined by weighing and measuring the volume of soil column. The porosity was 48 ± 4 percent, and the density was 0.88 ± 0.03 gm/cc (55 ± 2 lbs/cubic ft). The pore volume of the column was 11 ± 1 cubic centimeter (0.00039 cubic feet). The errors involved in this crude study are obvious, such as the disturbance and sieving of the soil, and the rough estimate of porosity and density. For an actual plan of discharge it would probably be necessary to obtain cores of the soil; ASTM methods would be used to measure the porosity and density of the core material.

The test solution was made up to simulate the aqueous discharge from a metallurgical flotation mill. The aqueous matrix was a saturated solution of gypsum, initially spiked with 1.0 ppm (mg/l) dissolved copper and 2.1 ppm (mg/l) dissolved molybdenum. The initial concentration of copper was made considerably higher than might be expected in an actual mill effluent; this was done because initial tests indicated that the soil had a large capacity for removal of copper from the test solution. The initial solution and the aliquots of effluent were analyzed for copper and molybdenum by atomic absorption and a thiocyanate colorimetric procedure, respectively. Standard additions were employed to test for sensitivity and interference in the copper analysis.

The results for molybdenum are shown in Figure 3. The results for copper are not shown because copper was generally below the limits of detection in the effluent during most phases of the study (the detection limit for dissolved copper was 0.01 parts per million).

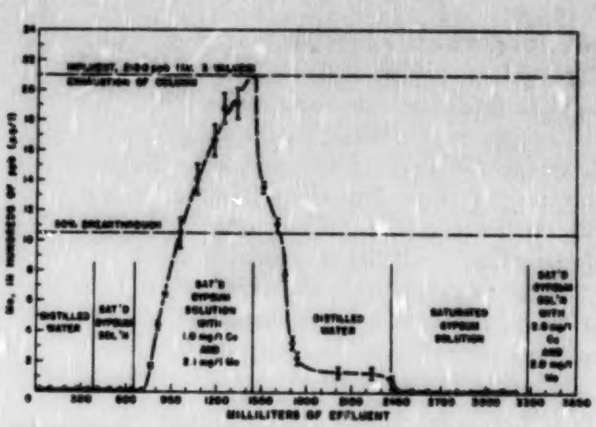

Fig. 3. Results of experimental study of leaching, removal, and remobilization of molybdenum using synthetic mill water and a New Mexico soil. Analytical error shown by vertical bars; error is approximately the diameter of the circles along the abscissa.

As shown in Figure 3, an initial period of leaching by distilled water (simulated rain water) failed to release detectable amounts of molybdenum (less than 15 parts per billion dissolved). The concentration of dissolved copper was approximately 0.05 ppm in the effluent during this initial leaching, which continued for 378 ml (34 pore volumes). This initial period of leaching shows that neither the concentration of copper nor molybdenum released by heavy applications of fresh water to Sulfur Springs soil poses a threat to ground water. Next, in Figure 3, the column was leached with approximately 26 pore volumes (288 ml) of metal-free synthetic mill water. In this case, dissolved copper in the effluent averaged less than 0.01 ppm and molybdenum less than 15 ppb. Again, these results show that extensive leaching with metal-free mill water fails to leach significant quantities of either metal from the soil.

At a volume of 661 ml in Figure 3 the metal-spiked water was introduced into the column, and 72 pore volumes were passed through. No detectable copper (less than 0.01 ppm) ever appeared in the effluent. Molybdenum was first detected after 66 ml (6 pore volumes) of spiked water had flowed through the soil. Fifty percent breakthrough of molybdenum occurred after passage of 32 pore volumes. The capacity of the soil to remove molybdenum was completely exhausted after passage of 72 pore volumes of mill water (792 ml). The difference in behavior of these two metals is caused by their ionic form in solution; molybdenum occurs as an anion (MoO_4^{--}), whereas copper is present as a cation (Cu^{++}) under these conditions. This soil had a very large capacity for removal of copper.

In Figure 3, after exhaustion of the column for molybdenum, the soil was again leached with simulated rainfall and metal-free mill water to test for desorption and remobilization. Of the 792 micrograms of copper introduced, leaching by 165 pore volumes of distilled water and synthetic mill solution released only about 19 micrograms; this means that the removal of copper by this soil approaches being irreversible. The initial leaching of the soil, prior to the introduction of the metal-rich mill water, released as much copper. In contrast, leaching of the loaded column with 83 pore volumes of distilled water remobilized nearly 60 percent of the approximately 700 micrograms of molybdenum previously added. The rate of release of molybdenum was decreasing exponentially with continued leaching by the distilled water, as shown in Figure 3, indicating that a very large volume of water would be required to remove all of the added molybdenum. Interestingly, upon initiation of leaching of the loaded soil with metal-free mill water, the concentration of molybdenum in the effluent fell to about 10 parts per billion, continuing to the end of this portion of the experiment with an additional 82 pore volumes (about 3280 ml on Figure 3). Calculations suggest that this decrease in desorbed molybdenum was probably due to the precipitation of solid $CaMoO_4$ in the soil due to the high concentration of dissolved calcium in the water.

Finally, in Figure 3, a new spiked mill water, containing 2 ppm copper was passed through the column in an attempt to cause breakthrough of the dissolved copper; however, breakthrough did not occur, even after a calculated addition of 256 ppm copper to the solid phases in the soil column.

The results of this study show that the soil from Sulfur Springs, New Mexico, has an enormously high capacity to remove copper, and that the removal is essentially irreversible. A relatively high capacity for removal of molybdenum is also exhibited, with 50 percent breakthrough at 32 pore volumes of spiked mill water. This type of information would comprise an important portion of a rational plan of discharge.

Space does not permit development here of expressions to relate the data of Figure 3 to a real situation. However, it can be shown from considerations of the distribution of the ions between the liquid and solid phases (Hajek, 1969; Tamura, 1972; Wierenga and others, 1975) that thousands of years may be required for the molybdenum in this study to reach an assumed water table at a depth of 30 meters (98 feet) in the Sulfur Springs soil, depending on the rate of discharge and infiltration. Copper would require much longer to reach the ground water.

SUMMARY

In summary, I have tried to demonstrate that there are valid chemical and physical reasons for looking toward the subsurface, especially the vadose zone in arid regions, for disposal of some types of wastes. Processes that operate in this environment can, under favorable circumstances, greatly attenuate and detoxify some wastes. On the other hand, we know of too many examples of contamination of ground water to believe that the overlying mantle of soil and sediment affords absolute protection. With these facts in mind, we must try to balance the need for protection of ground water against the economic realities of the use of water by industry, municipalities, and the general public.

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DISCUSSION

The following questions were answered by Donald D. Runnells after delivering his talk entitled "Wastewaters in the Vadose Zone of Arid Regions: Geochemical Interactions."

Q. by Leonard Wood. *Rain water is not distilled water. Have you used rain water in your column experiments?*
A. No, I have not tried to exactly duplicate the chemistry of rain water in the column studies. However, I do not believe that the observed variations in the chemistry of rain water would make a great deal of difference under most circumstances. The exception might be in regions of serious air pollution in which sulfuric acid becomes the principal control on the pH of rain water.

Q. by Mike Kaczmarek. *Would substantial increases in the Mo loading in the soil column result in comparable increases of Mo in the leachate resulting from "rain water" (distilled water)?*
A. Yes, the greater the initial loading, the greater would be the slug of Mo released from a soil when remobilization due to leaching does take place.

Q. by E. E. Jones, Dale Ralston and Harry Nightingale. *What percent of pore space was saturated in your experiments?*
A. This particular column was filled from the bottom to drive out air, then maintained in a fully saturated condition, so the pores were nearly 100 percent filled with water.

This was done because a limited amount of time was available and I had to maintain high rates of flow. A different experiment could be conducted under non-saturated conditions, in which the rate of flow would be substantially slower.

Q. by Harold Meiser. *What experiments are you planning for the future?*
A. We now have underway a program to test for leaching of trace elements from spent oil shale, and the reactivity of these elements with soils of the oil shale region in Colorado.

Q. by Neil Jaquet. *How do the volumes of distilled water used in your column studies compare to rates of precipitation in the area of concern?*
A. Experiments such as I described are designed to yield basic information, such as the distribution coefficient, K_d. Such information can then be used in equations involving the specific rates of precipitation and infiltration for an area. Thus, it is not necessary to model the *amount* of rainfall and infiltration, just the chemistry and rates of flow.

Q. by Jon O. Nowlin. *Assuming contaminant "removal" by cropping, how is the crop to be disposed of?*
A. The three advantages in cropping would be: (1) to convert the dissolved contaminants into solid form, or (2) to consume and transform such contaminants as nitrate, BOD, etc., or (3) to concentrate the contaminants from a large amount of fluid into a smaller volume of crop. However, in some cases, cropping would still offer no advantages because you still could not dispose of it.

Q. by K. E. Childs. *Can these tests be used to evaluate other areas, soils, and chemicals, or are the results too qualitative for this?*
A. The results are quantitatively useful for the particular parameters chosen, but new tests would have to be run for other discharges, soils, etc.

Q. by P. K. Saint. *Can one quantify the sorption capacity of the soil and predict the time of passage of an ion through the vadose zone? How about desorption later on?*
A. Yes, the results can be used for predicting migration, and references are given in the paper to illustrate this. Figure 3 in the paper shows the results of desorption studies.

Q. by Logan Kuiper. *What harmful effects would result if the contaminants reached ground water in the case considered?*
A. Molybdenum at this concentration (2.1 mg/l) could be toxic to ruminant animals, and the TDS (saturated gypsum solution) would degrade the quality of the ground water. Finally, the 1.0 mg/l copper would probably not be harmful to health, but would be undesirable because of taste.

Q. *How do different types of soils affect the leaching process?*
A. The processes of removal and leaching are strongly dependent on the type of soil.

Q. by G. F. Hendricks. *Any study of pick-up of copper by plants?*
A. There have been several such studies by others in the past, but this was not part of our program.

Q. by Don Lundy. *What mechanism retained the copper?*
A. I really did not get into the study of specific mechanisms. The experiments were designed only to give a general indication of capacity for removal and retention.

Q. by Buck Steingraber. *What happened to the sulfate in your percolating water—did it come through in the original concentration?*
A. The specific conductance that was measured in the effluent suggested that the sulfate passed through the soil with little or no removal.

Q. *What time equivalency in years of precipitation would be required to equal the amount of solution put through the soil column?*
A. For a semi-arid region, with perhaps 2 centimeters of recharge per year, the total volume of solution percolating through each cubic centimeter of soil would correspond to about 10 saturated pore volumes (assuming 20% porosity) per year. In this study I passed about 300 pore volumes through the saturated column, corresponding to 30 years of recharge. However, the ions in the experiment were retarded by the soil and would move much more slowly than the water.

Zoning Aquifers for Tertiary Treatment of Wastewater[a]

by Herman Bouwer[b]

ABSTRACT

Soils and aquifers can function as effective and economical filter systems for advanced treatment of conventionally treated sewage and other wastewater. The wastewater is applied to the land with low-rate or high-rate infiltration systems. Physical, chemical, and biological processes in the soil improve the quality of the wastewater as it percolates through the vadose zone and into the aquifer to become renovated water. The quality of the renovated water, however, often is not as good as that of the native ground water. To utilize the land for treatment of wastewater, without trading a problem of surface-water pollution for one of ground-water contamination, the spread of renovated water in the aquifer must be restricted. This can be accomplished by locating the system so that the renovated water drains naturally into a stream or other surface water, or by artificially removing renovated water from the aquifer with wells or drains at some distance from the application area. Examples are given of various systems that utilize these principles, and general design criteria are presented. Proper design involves analysis of underground-flow systems for various system geometries. Methods for measuring hydraulic conductivity, particularly in the vadose zone, are briefly reviewed.

INTRODUCTION

Land application of wastewater in which the pollutants are primarily of organic origin, like conventionally treated sewage and effluents from agricultural processing plants, can be an attractive alternative to additional in-plant treatment. Interest in land application of wastewater is increasing because of:

1. Legal restrictions on the discharge of wastewater into streams, lakes, or other surface water (Clean Water Bill, PL 92-500).

2. Favorable economic aspects, high reliability, and lower energy requirements than comparable in-plant treatment (including advanced techniques).

3. Opportunity for utilization of wastewater for irrigation of crops and use of nitrogen, phosphorus, and other "pollutants" in the water as fertilizer.

4. Opportunity for filtration of wastewater through soils and aquifers to produce renovated water for ground-water recharge. This water can then be reused for unrestricted irrigation, recreation, and other purposes.

TYPES OF SYSTEMS

There are three main types of land application systems: overland flow, low-rate infiltration systems, and high-rate infiltration systems. Overland-flow systems are used when soils are tight and infiltration is slow. Improvement in water quality is obtained primarily by flowing the wastewater in a shallow sheet over a soil surface covered by grass or other dense vegetation. Because of the low infiltration rates, overland-flow systems have little or no effect on the underlying ground water. With low-rate or high-rate systems, the wastewater infiltrates into the soil and the water-quality improvement is obtained as the water moves through the soil and down to the ground water. In low-rate systems, about 2 to 15 cm (1 to 6 inches) of wastewater are applied weekly or every 2 weeks. All systems that utilize wastewater for irrigation fall in this category. The water may be applied with surface-irrigation techniques (borders, furrows) or with sprinklers.

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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Surface-irrigation techniques require relatively smooth land, but they require less pumping (energy) and cause less air pollution than sprinkler systems. With high-rate systems, application rates vary from about 50 to 500 cm (2 to 20 ft) per week. Such amounts can be applied by over-irrigating agricultural crops or with special recharge or infiltration basins that generally are not vegetated. Intermittent flooding is the key to successful operation of rapid infiltration basins. Schedules range from 8-hour flooding and 16-hour drying each day to flooding and drying periods of several weeks each.

QUALITY IMPROVEMENT

The movement of wastewater through soil greatly improves its quality through physical, chemical, and biological reactions. Suspended solids are essentially completely filtered out. Bacterial activity in the soil results in almost complete removal of the biodegradable waste matter in the water, as expressed by the biochemical oxygen demand (BOD). However, not all organic compounds are biodegradable and some refractory organics may be present in the renovated water. For sewage effluent, for example, the renovated water may contain about 5 mg/l refractory organics, expressed as total organic carbon. The numbers of bacteria and viruses in the wastewater are greatly reduced, and often completely removed, as the water percolates through soil and aquifer(s).

For low-rate systems, most of the nitrogen in the wastewater is absorbed by crops, which can remove almost 100% of the nitrogen. For high-rate systems, the nitrogen is removed primarily by denitrification in the soil, which can remove about 80% nitrogen and more from secondary sewage effluent, as shown in laboratory studies (Lance *et al.*, 1976). Similar removal rates should be possible for field systems under favorable conditions and with careful management. Otherwise, lower removal percentages can be expected. Phosphates are also taken up by crops or they can be immobilized in the soil, particularly when the soil contains iron and aluminum oxides or is calcareous and alkaline. Phosphates are more mobile in sands with low pH. Immobilization of metal ions in soil is stimulated by pH values greater than 7 and by certain types of clay. Some metal ions, however, may remain in the water and move over long distances, especially when kept in solution by refractory organics that act as chelating agents. For a more detailed discussion of the various physical, chemical, and biological reactions in the soil and the chemical composition of different types of wastewater and renovated water, reference is made to Bouwer and Chaney (1974).

Low-rate (irrigation) systems in dry climates yield deep percolation water with a salt content that is about 3 to 10 times that of the original wastewater. Thus, unless the original wastewater has a very low salt content, the deep percolation water from those systems cannot be reused and must be handled like deep percolation or return flow from normal irrigated fields, including removal by drainage and "disposal" into surface water. For high-rate systems, the salt content of the renovated water is about the same as that of the original wastewater.

High-rate systems require relatively deep, light soils, preferably in the sandy-loam to loamy-sand range. Low-rate systems can be used on a wider range of soil textures. Coarse sands and gravels should be avoided. If such materials are also relatively shallow and underlain by fractured or cavernous rock, wastewater could penetrate to ground water in almost unchanged condition. Most of the quality improvement of wastewater in both low- and high-rate systems occurs in the top 1 m (3 ft) of soil. However, additional underground travel and underground detention time should be allowed for "polishing" treatment of the wastewater, including taste-and-odor removal and die-off of microorganisms (Gerba *et al.*, 1975).

ZONING OF AQUIFERS

While the wastewater is greatly improved in quality as it percolates through the soil in well-managed land application systems on properly selected sites, the quality of the resulting renovated water may not be as good as that of the local native ground water, whose quality is thus degraded. To take advantage of the beneficial aspects of land application without trading a surface-water pollution problem for a ground-water contamination situation, the spread of renovated water in the aquifer must be restricted. This can be accomplished by removing the renovated water from the aquifer at some point away from the application area. Sometimes this can be done by locating the land-application system so that the renovated water drains naturally into surface water (Figure 1). Where such a system is not possible, the renovated water can be collected at some distance from the application system by drains if the aquifer is shallow (Figure 2), and wells if the aquifer is deep (Figure 3). The portion of the aquifer between

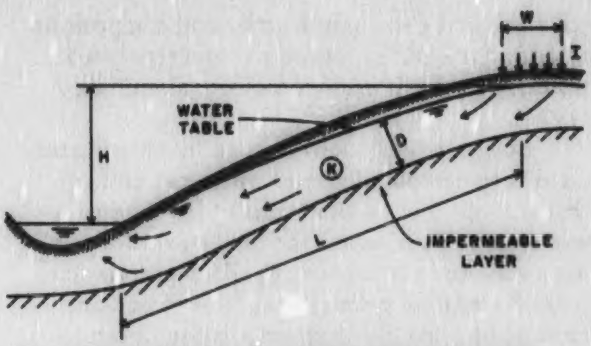

Fig. 1. Natural drainage of renovated water into surface water.

the application system and the discharge point of renovated water is then used as a natural filter system for renovating wastewater. Such zoning of part of an aquifer as a tertiary treatment facility for wastewater is not new. For example, if a certain separation is specified between a well and a septic tank, cemetery, or landfill, the portion of the aquifer in the zone of separation is in fact dedicated to treatment and dilution of the leachate from the pollution source. The difference between this conventional zoning and zoning for land-application systems is that, for the latter, the wastewater is removed from the aquifer after it has become renovated water. The renovated water may be discharged into surface water (through natural drainage or by pumping the discharge from the drains or wells into surface water) for indirect reuse, or it may be directly reused for unrestricted irrigation, unrestricted recreation, and certain industrial applications. Using renovated water for drinking is not yet encouraged, because the possible health effects of the refractory organics and other compounds present in low concentrations in the renovated water are not yet completely understood.

Examples of land-application systems where renovated water drains naturally into a stream are the high-rate infiltration systems of Fort Devens, Massachusetts, and Bielefeld, Germany. The wastewater for both systems consists of treated sewage effluent. Renovated water is collected by horizontal drains for the rapid infiltration systems in the dunes of western Holland, where polluted Rhine water is pretreated for municipal water supply, and at Dortmund, Germany, where a similar system is used to pretreat relatively high-quality surface water from the river Ruhr. An open ditch or trench is used at the Santee, California, system to intercept renovated sewage water for use in recreational lakes. Renovated water is collected with wells in the high-rate infiltration systems for Phoenix, Arizona, to enable reuse of the sewage water for unrestricted irrigation and recreation. Restricting the spread of renovated water in the aquifer by drains or wells is easier and less costly for high-rate than for low-rate application systems, because the former require much less land area than low-rate systems of the same capacity.

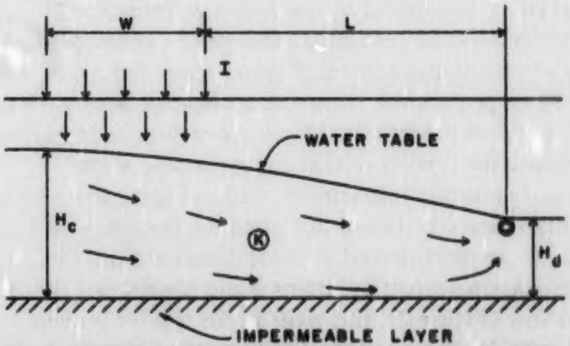

Fig. 2. Collection of renovated water by horizontal drain.

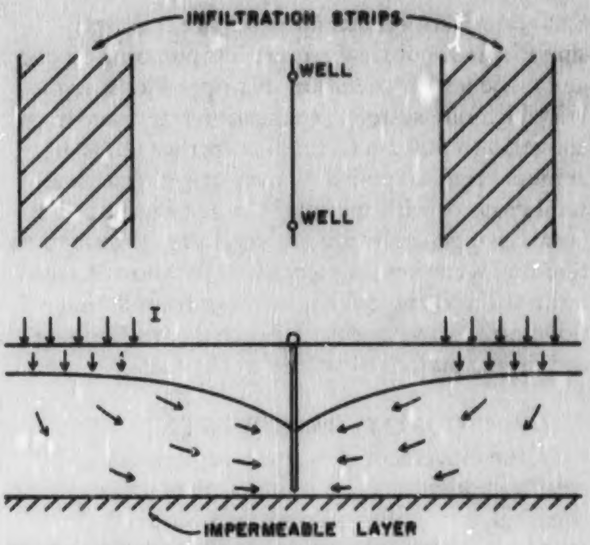

Fig. 3. Collection of renovated water by wells.

DESIGN

The design of a system for renovating wastewater by land application and filtration through soils and aquifers, and discharging the renovated water from the aquifer at some distance from the application system, should be based on the following criteria:

1. Ground-water mounds (including perched mounds) must not rise so high during application of wastewater that they restrict the infiltration rate. This can be accomplished by keeping the top of

the capillary fringe above the ground-water mound at least 0.5 m (2 ft) below the surface of the soil or the bottom of the infiltration basin [the corresponding water-table depth will then be about 1 to 2 m (3 to 7 ft), depending on soil texture]. Ground-water mounds should drop to a depth of at least 1.5 m (5 ft) within 2 or 3 days after cessation of wastewater application to permit sufficient aeration of the soil during periods of no infiltration (drying or resting).

2. The design lateral flow in the aquifer between the area of application of wastewater and that of discharge of renovated water should not exceed the flow that the aquifer can handle on the basis of its transmissivity and hydraulic gradients (this applies primarily to systems as in Figure 1).

3. The water must have traveled a sufficient distance underground and must have had sufficient underground detention time before it leaves the aquifer as renovated water. Desired distances and times of underground travel depend on the wastewater characteristics, the nature of the soil and aquifer materials, and the desired quality of the renovated water. For several systems, underground travel distances and detention times are about 100 m (300 ft) and 1 month, respectively (see also Gerba *et al.*, 1975).

Where renovated water discharges naturally into surface water, the lateral flow from the application area to the surface water can be restricted by insufficient aquifer transmissivity. In that case, the product WI of width of infiltration area and infiltration rate must not exceed the product KD H/L of aquifer transmissivity and slope of the water table (Figure 1). When KD H/L is relatively small, the wastewater-application area should be long and narrow so that W and, hence, IW are also small.

Where the renovated water is collected by underground drains (Figure 2), the flow system below the water table can be described by the following equation (Bouwer, 1974):

$$H_c^2 = H_d^2 + IW(W + 2L)/K \qquad (1)$$

where H_c = maximum height of water table above impermeable layer,

H_d = height of water table above impermeable layer at drains (height of center of drains if drains are running free with no back pressure),

I = infiltration rate (average for entire area of width W),

W = width of application area,

K = hydraulic conductivity of aquifer, and

L = distance between edge of infiltration area and drain.

Equation 1 is based on the assumption of horizontal flow, which is valid if H_c is relatively small as compared with W + L. The value H_c should be taken at the outer edge of the infiltration area if the drain is only on one side of the area. However, if drains are on both sides of the infiltration area, H_c refers to the water table below the center of the application strip and W represents one-half the width of the application area. Equation 1 can be used to calculate L for various combinations of W, I, and H_c and H_d so that the best geometry of application area and drain location can be selected. The transient situation, i.e. predicting the rise and fall of ground-water mounds in relation to time, can be handled with equations developed by Marino (1974a and 1974b) for recharge areas bound on one or both sides by surface water with constant water level.

For deep aquifers, the renovated water is more effectively intercepted by wells than by horizontal drains. If several such wells are uniformly spaced on a line midway between two parallel infiltration strips (Figure 3), it is theoretically possible to collect essentially all the water that entered the soil as wastewater. To achieve this, the wells should completely penetrate the aquifer (or at least the upper, active region if the aquifer is deep enough to develop active and passive regions). In addition, infiltration and pumping rates should be managed so that ground-water levels below the outer edges of the infiltration strips are not affected by the infiltration-and-pumping system and remain at the same level as the ground water in the aquifer adjacent to the renovation system. For this purpose, observation wells for monitoring ground-water levels should be installed at the periphery of the system.

To obtain the optimum design of a system such as in Figure 3 (width and length of infiltration strips, distance between strips, distance between wells, and capacity of wells), the water-table drop from the outer edge of the recharge system to the wells should be calculated for various designs so that the best system geometry can be selected. This water-table drop can be predicted by superimposing the ground-water mound formed by infiltration from the recharge strips on the drawdown due to the pumped wells. The rise and fall of ground-water mounds in aquifers of large lateral extent can be predicted with equations or graphs presented by Hantush (1967), Bianchi and Muckel (1970), Hunt (1971), and Singh (1972). Drawdowns around wells can be calculated with conventional well-flow theory. Steady-state solutions for the flow system between the recharge areas and the wells for the geometry of Figure 3 were presented by Bouwer (1970). Equations developed on the basis of horizontal-flow theory should be used with caution where the thickness of the aquifer is larger than the width of the infiltration area, because the lower portions of the aquifer then do not contribute much to the recharge flow system and are essentially stagnant. In those cases, the effective transmissivity of the aquifer for recharge is less than the total transmissivity of the aquifer (Bouwer, 1962).

MEASUREMENT OF HYDRAULIC CONDUCTIVITY

To predict underground-flow systems in connection with land-application systems for wastewater with discharge of renovated water from the aquifer, values of the hydraulic conductivity K in the aquifer and the vadose zone must be known. Such knowledge of K also enables prediction of maximum infiltration rates. Local experimentation, however, is usually required to determine the effect of clogging, biological activity, and weather on infiltration rates, so that a realistic estimate of the infiltration rates can be obtained.

Values of K should preferably be measured in place. The K-values of aquifers can be obtained with pumped-well techniques, like the Theis pumping test, or with the slug test where a volume of water is suddenly removed from a well and the subsequent rise of the water level in the well is measured for calculation of K. The theory for the slug test, which originally was developed for completely penetrating wells in confined aquifers (Cooper, *et al.*, 1967) has recently been extended to partially penetrating wells in unconfined aquifers (Bouwer and Rice, 1976). Point measurements of K, as may be required to detect layers of different K in the aquifer, can be obtained with piezometer techniques (Bouwer and Jackson, 1974, and references therein). For the vadose zone, K may have to be determined in vertical direction to predict effects of restricting layers on perching mounds and to estimate potential infiltration rates. There may also be cases where K of the vadose zone should be measured in horizontal direction, for example, if infiltration will cause the water table to rise and part of the vadose zone will become saturated and contribute to lateral flow of ground water.

The hydraulic conductivity in vertical direction in the vadose zone can be determined with the air-entry permeameter and infiltration-gradient techniques (Bouwer and Jackson, 1974, and references therein). The air-entry permeameter (Figure 4) is essentially a covered infiltrometer with standpipe and reservoir to let the water infiltrate into the soil at high head. The infiltration rate is calculated from the rate of fall of the water level in the reservoir. When the wet front is expected to have reached a depth of about 10 cm (4 inches, equal to the depth of penetration of the cylinder), a valve at the base of the pipeline connecting the reservoir to the cylinder is closed. This halts infiltration and causes the pressure of the water inside the cylinder to decrease until the air-entry value of the soil above the wetting front is reached, after which the pressure of the water inside the cylinder slightly increases. Measuring the minimum water pressure inside the cylinder with, for example, a vacuum gauge with memory pointer, enables the air-entry value at the wetting front to be evaluated. This value is then used to estimate the pressure head at the wetting front while water was still infiltrating. As soon as minimum pressure is detected, the air-entry

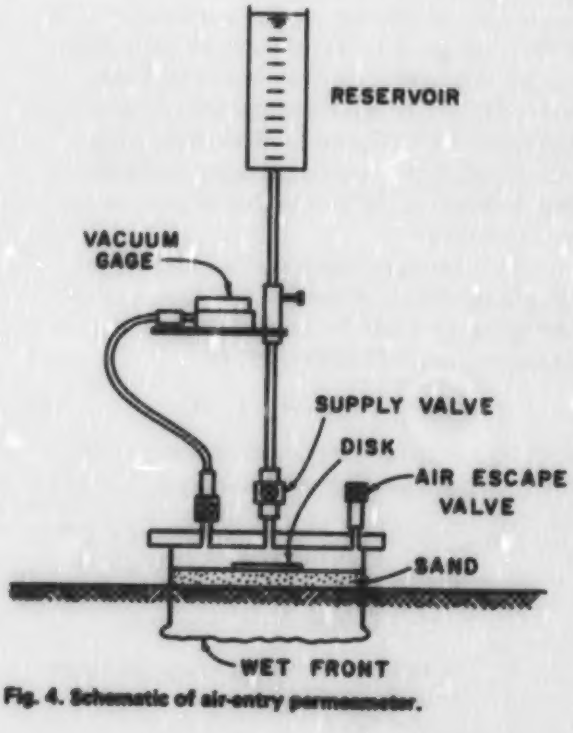

Fig. 4. Schematic of air-entry permeameter.

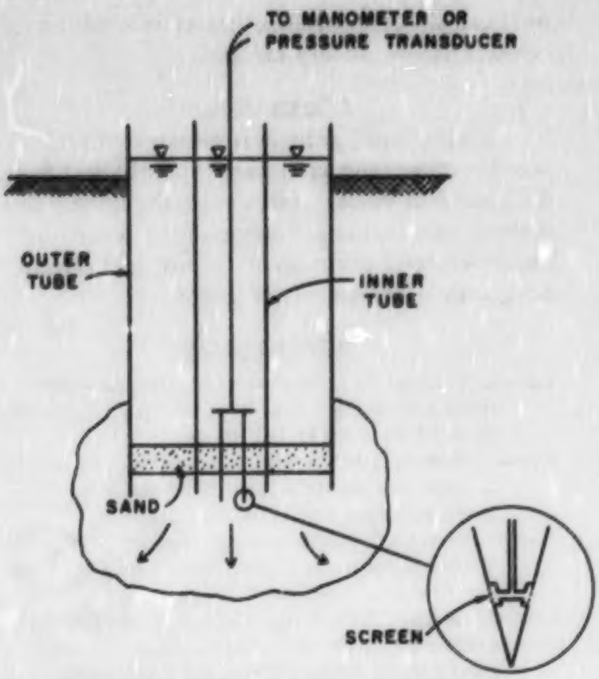

Fig. 5. Schematic of infiltration-gradient method.

permeameter is removed and the depth of the wetting front is measured, for example, by digging with a spade and observing how deep the soil was wetted. Knowing the depth of the water above the soil, the infiltration rate just before closing the valve, the depth of the wetting front at the time the valve was closed, and the pressure head at the wetting front while it was advancing downward, K in vertical direction of the wetted zone can be calculated with Darcy's equation. The method requires about 10 liters (3 gal) of water per test and is usually completed in less than 0.5 hour. In its present form of construction, the air-entry permeameter is a surface device. To use the device for measuring K at a certain depth, pits or trenches must be dug.

With the infiltration-gradient technique (Bouwer and Jackson, 1974, and references therein), an auger hole is dug and lined with a steel tube. A second steel tube of smaller diameter is placed concentrically in the hole and pushed or driven about 5 cm (2 inches) into the hole bottom (Figure 5). The tubes are filled with water and constant, equal, water levels are maintained in both tubes. Infiltration then creates a wetted zone with positive pressures in the soil below the auger hole. The infiltration rate from the inner tube is measured while small, fast-reacting piezometers are pushed into the auger-hole bottom at increments of about 2 cm (1 inch) to determine the vertical hydraulic gradient in the wetted zone. The K-value in vertical direction below the hole bottom then can be calculated with Darcy's equation. Once the hole is dug and the tubes are installed and filled with water, the test can usually be completed in about 1 hour, depending on K.

A third technique for measuring K in vadose zones is the double-tube method (Bouwer and Jackson, 1974, and references therein), which also consists of two concentric tubes in an auger hole with the inner tube penetrating the hole bottom about 2 cm (1 inch). The tubes are covered with a lid that has two standpipes—one connected to the inner tube and the other to the annular space between the inner and the outer tube in the hole (Figure 6). The tubes are filled with water and water levels are maintained at the top of the standpipes to create a wetted soil region with positive water pressures below the auger hole. When this region has reached sufficient size, the inflow into the inner tube is stopped and the fall of the water level in the inner-tube standpipe is measured. This is done while keeping the outer-tube standpipe full and, in another measurement, while adjusting the water level in the outer-tube

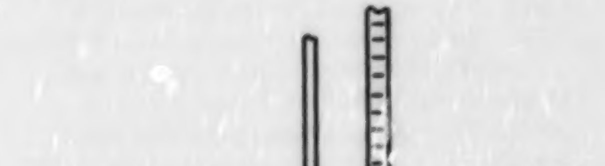

Fig. 6. Schematic of double-tube method.

standpipe so that it falls at the same rate as that in the inner tube. These measurements then yield the reduction in infiltration from the inner tube when the pressure head in the inner tube is less than that in the outer tube, which enables calculation of K of the wetted zone. The resulting value is affected by K in vertical and horizontal direction, but it mostly reflects K in vertical direction. The double-tube method requires about 2 to 5 hours to complete and about 100 liters (30 gal) of water per test.

Values of K in horizontal direction in the vadose zone can be determined with the shallow-well pump-in method. This technique consists of drilling an auger hole, filling it with water, and maintaining a certain water level in the hole until the outflow from the hole has become constant (Bouwer and Jackson, 1974, and references therein). Several days may be required before the outflow approaches a constant value and large quantities of water (hundreds of liters or about 100 gal) may be necessary. The value of K is calculated from the constant outflow rate and the hole geometry. The resulting value primarily reflects K in horizontal direction. A true measure of K in horizontal direction can be obtained by combining the double-tube method with the infiltration-gradient method in the same hole (Bouwer and Jackson, 1974, and references therein).

Where infiltration basins or other facilities are already available, like an experimental or pilot ground-water recharge system, the transmissivity, T, of the aquifer can also be determined from the measured rate of rise of the ground-water mound in response to infiltration (Bianchi and Muckel, 1970). This procedure yields the ratio T/f, where f is the fillable porosity. If f is known (f can be evaluated as the difference between the volumetric water contents above and below a rising ground-water mound), T can be calculated. When using this procedure, care should be taken in case T is measured from the mound rise below a narrow recharge strip and then used to predict the mound rise below a much wider strip or basin. This is because recharge flow systems in aquifers have an upper, active zone and a lower, passive zone if the height of the aquifer is larger than the width of the recharge basin (Bouwer, 1962). Thus, the effective transmissivity for a narrow recharge strip could be less than the actual transmissivity for the entire aquifer, and less than the effective transmissivity for a wider recharge strip. This would overestimate the height of a ground-water mound below a wide infiltration area if the height is based on T calculated from the observed mound rise below a narrow infiltration area.

CONCLUSION

Zoning aquifers for wastewater renovation is possible if the renovated water is discharged from the aquifer at some distance from the infiltration system. Successful application of the technique requires careful selection of the site, and careful design and operation of the system.

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DISCUSSION

The following questions were answered by Herman Bouwer after delivering his talk entitled "Zoning Aquifers for Tertiary Treatment of Wastewater."

Q. by Jeffrey A. Gilman, William A. Trippet II, Randy Sweet, J. E. Biesecker, and William H. Walker. *Is there any trace metal contamination of the aquifer associated with spreading of the sewage sludge in the Bielefeld, Germany project?*
A. I do not know of situations where heavy metals in sewage effluent had adverse effects on crops or soils in land treatment or irrigation systems. Should such problems develop, applications of lime may be effective in immobilizing the metals in the soil. Source control of metals would be the logical approach where heavy-metal concentrations in sewage effluent are unusually high. Most of the metals in sewage end up in the sludge. Unless metal concentrations are abnormally high, metal problems in soils and crops are not anticipated if sludge applications are based on crop nitrogen requirements.

Q. by Jack Robertson. *Has the movement of any organic constituents been measured in the Phoenix project?*
A. BOD is reduced to essentially [illegible] in the renovated water. Refractory organics we[illegible]d present at levels of 4 to 7 mg/l total organic carbon.

Q. by P. K. Saint, Bill Chase, and Don Runnells. *What about the fate of stable organics and viruses during their travel through vadose zone and zone of saturation? Is there data from European experiences?*
A. The state of knowledge is incomplete but results so far indicate that movement and detention of viruses in soil depend on soil, climate, and ionic composition of water. Fine-textured soils with a low pH effectively remove viruses from sewage effluent with a relatively high TDS and concentration of divalent ions. Rainfall during drying can cause previously adsorbed viruses to move deeper into the soil. Application of calcium chloride to the soil prior to adding distilled water, however, prevented mobilization of viruses in laboratory studies. A detailed article on these phenomena will be published by J. C. Lance and coworkers in the October 1976 issue of *Applied and Environmental Microbiology*.

Q. by G. C. Slawson. *Are the wastewaters you are addressing chlorinated as part of the treatment before disposal? If so, is there not a potential for introducing chlorinated hydrocarbons into the environment? This would clearly have to be considered for water reuse planning.*
A. The Phoenix projects utilize unchlorinated secondary effluent. Chlorination is a waste of money if effluent is used for rapid-infiltration systems in soils and aquifers that remove all pathogens anyway.

Q. by W. B. Wilkinson. *If a rapid infiltration system recharges water with nitrate-nitrogen in excess of 10 mg/l to the water table—will the nitrate degrade as it moves through the saturated zone to a river or well? If not, should such systems be used?*
A. Denitrification requires organic carbon as energy source for the bacteria. Aquifers usually are devoid of readily available organic carbon. Hence, once nitrate has reached an aquifer, it is usually not attenuated other than by dispersion. Whether rapid infiltration should be used or not depends on how much nitrogen can be removed and where the renovated water will go. Under the proper hydrogeological conditions and with good management, it may be possible to remove one-half to two-thirds of the nitrogen from the sewage water. Since some of the nitrogen will remain as ammonium, this will bring the nitrates down to near or below the limit of 10 mg/l nitrate-N.

Q. by W. B. Wilkinson. *How can one be certain that smearing does not occur at the base of a permeability test auger hole—leading to false permeability test results?*
A. Smearing could be a problem for the reverse auger-hole method, also called the shallow-well pump-in technique. For the infiltration-gradient technique, smearing is no problem because the vertical gradient is measured directly. For the double-tube method, smearing is minimized by the use of a special hole cleaner which takes a bite out of the bottom and leaves an undisturbed surface with a clean break. For the air-entry permeameter technique, the surface is accessible and can be cleaned or brushed to expose a natural surface.

Q. by Jim Braithwaite. *What was the sample volume (per sample) in the virology analyses?*
A. For the renovated water, the target sample volume was 100 gallons. In practice, it varied from 46 to 120 gallons. This volume was passed through filters (using pH-adjustments) to concentrate the viruses. For secondary effluent, viruses were concentrated from 1- to 5-gallon samples.

Q. by N. Jaquet. *What percentage of consumptive use (via evaporation) is being experienced in the Phoenix system?*
A. Annual hydraulic loading rates are 200 ft to 300 ft. Annual evaporation is about 6 ft. Thus, 2 to 3% of the water is lost to evaporation.

Q. by G. F. Hendricks. *Would septic tank effluent absorption field be improved by phosphate removal?*
A. Not as a rule. Since phosphate precipitates as apatite-like compounds in some soils, the permeability could eventually become smaller. This will take a very long time, however. Where soils do not remove much phosphate (acid, sandy soils, for example) and the effluent seeps into a lake or other surface water, it may be desirable to precipitate phosphates so as to reduce the pollution danger of the surface water.

Q. by G. J. Thabaraj and Martin J. Allen. *What is your recommendation regarding the minimum distance that should be maintained between a low-rate or high-rate land application system (for secondary sewage effluents) and a potable water-supply well?*
A. Under favorable conditions (relatively deep and fine-textured soil, relatively deep water table, granular aquifer materials with no large pores, and scheduling drying periods

to avoid large rainfall just after start of drying), a separation distance of a few hundred feet between a potable-water-supply well and a land treatment system may be adequate, at least with respect to pathogenic microorganisms. The greater the distance, the better. Water from any well that may have the slightest chance of containing renovated sewage effluent should be routinely chlorinated or otherwise disinfected prior to drinking.

Q. by K. Childs and Ken Sylvester. *The soil zone beneath a disposal system is a fixed volume with a fixed capacity for waste assimilation. Many States require only 5 feet of aerated zone beneath sites. Doesn't this imply that each system has a fixed (limited) filter life? Does waste filtration take place in the saturated zone as well as the aerated zone?*
A. Most biological and other renewable renovation processes take place in the top few feet of the aerated zone. Thus, 5 feet will be adequate for most cases. Additional renovation will take place under saturated conditions, except in very coarse or fractured materials where the water flows through large pores or channels.

Q. by Pete Atkins. *You started your presentation by indicating that land disposal meets the goal of "zero discharge." This is not completely true unless the renovated water is completely recycled. Land disposal does not meet the goal of zero discharge. Do you agree?*
A. Land treatment meets the goal of zero discharge into surface water. If the wastewater eventually drains back into surface water via underground movement, it will do so as "renovated" water and cause much less pollution than the original wastewater.

Q. by Don Lundy. *Assuming you have unsaturated flow in the vadose zone below an infiltration pond, don't you need to adjust the vertical hydraulic conductivity determined in your saturated flow field test where you used Darcy's Law?*
A. The "saturated" hydraulic conductivity values of vadose zones yielded by field tests give an upper limit of infiltration rates that might be achieved. So if the average vertical hydraulic conductivity (harmonic mean!) is 0.5 m/day, we know that we cannot push more than 0.5 m/day into the ground. In actuality, infiltration rates will be less than the maximum limit because of clogging at the surface. In that case, the underlying material will be unsaturated with the unsaturated hydraulic conductivity numerically equal to the infiltration rate (assuming unit hydraulic gradient).

Q. by R. S. Domenowske. *What is your experience with the role of carbon nitrogen ratios on chemical and biological soil processes that might affect treatment capacity?*
A. C/N ratios in sewage effluent are important for denitrification, which requires about 1 mg organic carbon per mg nitrate nitrogen denitrified. Secondary effluent does not have enough organic carbon for complete denitrification for its nitrogen, particularly since some organic carbon will already be used up by the time ammonium has been converted to nitrate. Primary effluent has a more favorable C/N ratio and more denitrification has been obtained with primary effluent than with secondary effluent.

Q. *What kind and degree of treatment of the wastewater is required before putting it in the infiltration basins in order not to decrease the infiltration rate of the basins and hydraulic conductivity of the aquifer?*
A. Most sewage effluents will have had secondary treatment when used for rapid infiltration. Pre-sedimentation before letting the effluent flow into the infiltration basin may be necessary if suspended-solids concentrations are too high (for example, above 20 mg/l). Total detention time in pre-sedimentation and infiltration basins should be sufficiently short to prevent excessive growth of algae. Where the in-plant and land-treatment systems are to be built from scratch, it may be advantageous to apply primary treatment only and use well-settled primary effluent for infiltration. This may reduce infiltration rates, but the in-plant treatment would be much cheaper and denitrification in the soil would be stimulated because the primary effluent has more organic carbon.

Q. by George Clark and Steven Cordiviola. *Generally speaking, how long can a given plot of land be utilized effectively as a filter?*
A. For the renewable processes (removal of BOD, suspended solids, bacteria and viruses, denitrification), there will be essentially no end to the useful life of a land treatment system. Adsorptive processes, which can remove phosphate and metals, on the other hand, are effective over a limited time. Adsorbed phosphate, however, slowly reverts to insoluble forms like calcium phosphates. Metals can also be precipitated. The precipitates may eventually accumulate in soil pores and on soil particles to such a level that soil permeability is reduced. This will take a long time, however. Useful life of land treatment systems at carefully selected sites should be on the order of decades and possibly even centuries.

Q. by Mike Apgar. *In the humid German climates where high-rate infiltration was mentioned, what was the rate of "overapplication," its effect on crops, clogging problems, and TOC concentrations and/or specific organics identified in the renovated water?*
A. Annual application rates were 8 to 15 m. Crop effects must have been beneficial, because more farmers wanted to get in on the project. Quality data on the renovated water were not available.

Q. *How does the cost of the percolation-pumping scheme for Phoenix compare with that for chlorination and direct discharge to canals? What about energy consumption? How does percolation and pumping, etc. compare with other means of removing bacteriological constituents?*
A. Chlorination would probably be cheaper than renovation by rapid infiltration. However, chlorination of secondary effluent does not kill the viruses inside suspended solids. Also, BOD and suspended solids contents of the chlorinated effluent would be too high to meet State requirements for unrestricted irrigation. In addition, it is desirable to remove at least half of the nitrogen (to avoid adverse effects of high-nitrate irrigation water on crops), more than 90% of the phosphorus (to make the renovated water suitable for recreational lakes), and to make the water aesthetically acceptable for reuse in a densely populated area (renovated water from a land treatment system looks crystal clear and is free from taste and odor).

Q. by L. A. Swain. *Can you conclusively say that in long-term operation of the high-rate system the nitrate will not*

be taken out by the unsaturated zone up to a certain limit and be leached out with subsequent infiltration at a higher concentration than the inflowing water initially was?
A. Nitrates not denitrified during drying are flushed out by the newly infiltrating water when a new flooding period is started. Some of this nitrate may then be denitrified as it mixes with the sewage water. The remaining nitrate then moves downward with the renovated water and forms a nitrate peak. Maximum nitrate concentrations in this peak have exceeded total N concentrations in the sewage effluent. However, with additional underground movement and lateral flow, the peaks are attenuated to low "plateaus."

Q. by Roger Clissold. *What is the population of Bielefeld and what quantity of effluent is involved? Do you feel there is an upper practical size for such a system?*
A. The population of Bielefeld was 260,000 in 1967. Water use was 150 l per person, so that effluent production probably was around 35,000 m^3/day or about 10 mgd. The infiltration area initially was 50 ha (125 acres) with plans to expand it to 300 ha (750 acres). Generally land treatment systems are relatively small, but there are also some very large systems like the Melbourne, Australia, system (20,000 acres serving 2 million people) and the Muskegon County, Michigan, system (10,000 acres).

Q. by Roger Clissold. *Phoenix system – does the sand and gravel below the ponds become saturated from the bottom of the pond down to the original water-table level at any time?*
A. Saturation of the soil material below the infiltration basins is unlikely because infiltration is restricted by a clogged layer on the surface. The only way that saturation could occur is through perching layers.

Potential Replacement of Septic Tank Drain Fields by Artificial Marsh Wastewater Treatment Systems[a]

by C. W. Fetter, Jr.[b], W. E. Sloey[c] and F. L. Spangler[c]

ABSTRACT

Individual subsurface liquid waste disposal has been cited as a source of ground-water contamination. Wastewater treatment systems using emergent marsh vegetation planted in a gravel substrate in a plastic-lined trench could be used to treat septic tank effluent. A pilot plant treating unchlorinated primary municipal effluent achieved the following reductions in mass: BOD_5–77%; COD–71%; orthophosphate–35%; total phosphorus–37%; nitrate–22%; coliform bacteria–99.9%. While such treatment is possible only during the growing season, it could be useful at summer cottages, camping areas, resorts and roadside rest areas. Marsh treatment systems are inexpensive to operate and virtually automatic.

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INTRODUCTION

Research activities in problems of small-scale liquid waste disposal have tended to focus on either improving the efficiency of subsurface disposal systems or applying conventional treatment methods (Bouma and others, 1972). Our research has concentrated on developing a new method of wastewater treatment using emergent aquatic vegetation for the treatment of liquid domestic wastes. Such treatment could be useful in summer cottages, camping areas, resorts, and roadside rest areas.

GROUND-WATER POLLUTION FROM SEPTIC TANKS

The most common method of small-scale liquid waste disposal is via septic tanks and soil absorption drain fields or cesspools. The septic tank removes solids from the waste by settling and grease by flotation. Anaerobic biological decomposition of the settled sludge produces gas and releases soluble organic matter. The effluent is high in BOD, bacteria, organic and ammonia nitrogen, and phosphorus (Dudley and Stephenson, 1973). It is similar in composition to primary sewage effluent.

Soil absorption of effluent tends to remove organic matter and bacteria by filtration, forming a crust of organic matter, colloids and microorganisms below the seepage bed (Bouma and others, 1972). Phosphate is generally removed by adsorption on the soil or chemical precipitation (Syers and Armstrong, 1971). Nitrogen is basically conservative, with the ammonia and organic forms being converted to the nitrate form by nitrifying bacteria although under some conditions denitrification may occur (Lance, 1972).

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Bouma and others (1972) list several conditions under which soil absorption fields either fail or could not be constructed: Formation of a thick crust beneath the drain field which tends to partially seal it. The soil surrounding the seepage bed has a natural permeability which is too low. Bedrock, either impermeable or creviced, is too close to the surface for a drain field to be constructed. The water table is too close to the surface. In addition, soils which are too highly permeable (gravels) or chemically unreactive may also not properly remove wastewater constituents.

Under these conditions septic systems are either not feasible or may fail and result in ground-water contamination. There are numerous examples of ground-water contamination due to soil absorption field failure (e.g. Fetter, 1974). In many areas rural development is limited by poor soil conditions. In some Wisconsin counties, more than 95% of the land area is underlain by soil conditions which preclude the use of drain fields.

Holding tanks which are pumped and the effluent taken to a treatment plant are an alternative to soil absorption fields, but disposal by this method is quite expensive.

ARTIFICIAL MARSH TREATMENT SYSTEMS

Aquatic vegetation has been used for wastewater treatment in Europe with varying degrees of success (Seidel, 1966, 1971a, 1971b, 1976; Althaus, 1966; Kickuth, 1969; Kok, 1974; de Jong, 1976). However, it was not known if plant species found in the north-central United States could be used. Accordingly, our research initially focused on selection of suitable North American species. We also experimented with plant propagation methods and evaluated various substate materials in terms of suitability for plant growth.

As a result of these initial studies, we determined that the bulrush *Scirpus validus* could be propagated from rhizomes and grew well when planted in a shallow gravel substrate (Spangler, Sloey and Fetter, 1976a).

Experimental Facilities

An experimental facility was constructed in Seymour, Wisconsin (population 2,257) next to the municipal wastewater treatment plant. Artificial marshes were made by lining shallow trenches with PVC plastic, filling the trenches with gravel and hand planting vegetation. The experiments described in this paper were made in a pilot plant with a surface area of 85 m². It was in a trench 19.3 m long, 3.05 m wide at the bottom, and 5.8 m wide at the top. The trench had plywood ends, was lined with 20 mil PVC plastic, and filled with 15 cm of sand at the bottom, then 30 cm of coarse gravel and 30 cm of pea gravel on the top (Figure 1).

There were provisions for withdrawing either unchlorinated primary or secondary effluent from the municipal treatment plant, and adding metered amounts to the basin through a pipe which allowed a free fall of effluent.

The water from the artificial marsh was withdrawn from the gravel substrate via a drain tile across the lower end. Marsh effluent was also metered. A smaller basin which had an area of about 10 m² was used as a control. It had a gravel substrate but no emergent vegetation (Spangler, Sloey and Fetter, 1976b).

Several nests of small plastic pipes were inserted vertically into the gravel to depths of 15, 30, 45 and 60 cm. These enabled the withdrawal of water quality samples from various locations and levels in the pilot plant (Figure 1).

Experimental Methods

Unchlorinated primary effluent was applied to the pilot plant for the period August 21-November 4, 1975. The vegetation and microbiota of the marsh had been previously established by two seasons of experiments with secondary effluent.

Effluent was applied at an average rate of 2.5 m³/day which yields a loading factor of 29 l/m² of surface area and a retention time of about 10 days. Grab samples were collected once a week from the marsh outlet and from the small sampling wells in the gravel. A twenty-four hour composite sample of the marsh influent was taken by an automatic sampler. Prior experiments had shown through intensive sampling that while there

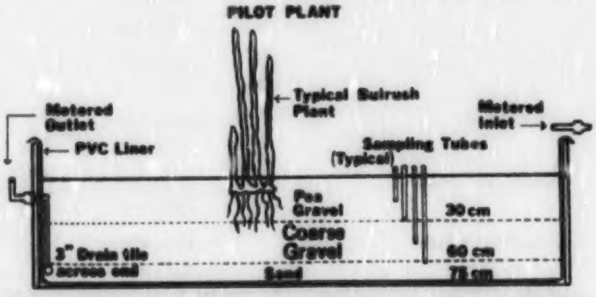

Fig. 1. Schematic diagram of artificial marsh pilot treatment plant. (Not to scale. Total length 19.3 m.)

was considerable chemical variation in the marsh influent (wastewater treatment plant effluent), there was little variation in the marsh effluent. Samples were refrigerated and analyzed within 24 hours of collection by Environmental Protection Agency (1971) methods.

Results

Summary results of the effectiveness of the pilot plant are given in Table 1. The values given are for concentrations of solute in mg l^{-1}. Tables 2 and 3 give the water quality of influent and effluent water respectively. During this period a total of 188.3 m^3 of water drained from the marsh. Evapotranspiration amounted to 37.9 m^3 during the period or 0.44 m depth during the 75-day period. The effect of evapotranspiration is to concentrate the solute in the marsh effluent. However, as rainfall was almost equal to evapotranspiration during this period, this tendency was counteracted by the dilution of the rainfall.

The following reductions in mass (corrected for evapotranspiration) were achieved: BOD$_5$–77%; COD–71%; orthophosphate–35%; total phosphorus–37%; nitrate–22%; coliform bacteria–99.9%.

The design of the pilot plant tended to reduce the effectiveness of the bulrush treatment. Much of the wastewater passed through the coarse gravel layer at the 30 to 60 cm depth. The bulrush roots did not generally extend to that depth. The bulrush exudes oxygen through its roots into the sediments surrounding them. In experimental basins with 15 cm gravel, there was always dissolved oxygen in the marsh effluent. In the pilot plant with the layer of coarse gravel, there was zero dissolved oxygen in the deeper gravel layers. However, the surface gravels were aerobic, and therefore no noxious odors developed. With a shallower gravel substrate, greater treatment would be anticipated since the sediments would be aerobic.

The deep layer of coarse gravel created chemical stratification in the gravel. Figure 2 shows the mean concentrations of BOD$_5$, COD, orthophosphate, and total phosphorus at various locations within the gravel. Concentrations are seen

Table 1. Effectiveness of Pilot Plant in Treating Primary Effluent[a] – Summer 1975

Parameter	*Source*	*Mean*[b]	*Std. Dev.*	*Number*	*Percent reduction*
BOD	Influent	317	133	10	–
	Effluent	72	36	9	77
COD	Influent	401	158	12	–
	Effluent	117	94	9	71
Orthophosphate	Influent	26	6	12	–
	Effluent	17	3	9	35
Total phosphorus	Influent	27	6	12	–
	Effluent	17	4	9	37
Conductivity (μmhos)	Influent	1330	174	12	–
	Effluent	1353	160	9	−2
Turbidity (JTU)	Influent	48	16	12	–
	Effluent	38	5	9	21
Nitrate N	Influent	0.27	0.27	10	–
	Effluent	0.21	0.09	9	22
Coliform (log col. 100 ml^{-1})	Influent	7.31	6.72	7	–
	Effluent	5.07	5.17	6	99.9
pH	Influent	7.28	0.12	12	–
	Effluent	7.27	0.23	9	–
Total solids	Influent	973	113	3	–
	Effluent	1156	19	2	−19
Suspended solids	Influent	42	47	3	–
	Effluent	32	14	2	23
Dissolved solids	Influent	932	84	3	–
	Effluent	1124	34	2	−21

[a] Mean values were not corrected for evapotranspiration.

[b] All values expressed as mg l^{-1} unless otherwise indicated in parentheses.

Table 2. Water Quality of Primary Sewage Added to the Pilot Plant

1975	*BOD*[a]	*COD*[a]	*O-P*[a]	*t-P*[a]	*Cond.*[b]	*Turb.*[c]	*Colif.*[d]	*NO*$_3$[a]	*pH*
26 August	>324	309.68	24.30	22.72	1080	44		1.00	7.15
28 August	>154	354.42	23.33	25.07	1260	43		.04	7.25
4 September	250	172.48	12.0	12.52	1310	26	7.29	0.22	7.40
9 September	295	328.6	31.32	29.52	1060	39	7.204	0.22	7.15
16 September	245	284.2	28.61	30.99	1440	36		0.34	7.25
24 September	393	791.0	28.86	27.50	1450	80		0.10	7.50
30 September	255	395.5	31.87	31.08	1425	39	7.267	0.24	7.20
7 October	480	309.6	22.84	24.04	1200	31	7.505	0.21	7.2
15 October	345	362.8	31.97	32.42	1300	42.5	7.301		7.2
21 October	570	554.8	30.48	32.19	1600	55.5	7.301	0.25	7.4
28 October	320	487.3	24.11	27.99	1180	67	7.27	0.00	7.4
4 November	215	457.84	26.01	30.44	1550	67		0.36	7.25
Mean	316.8	401.52	26.32	27.21	1330	47.5	7.314	0.271	7.28
Standard Deviation	133.48	157.94	5.64	5.63	174	16.37	6.715	0.267	0.12

[a] concentrations in mg l^{-1}
[b] micromhos
[c] J.T.U.
[d] log units

Table 3. Water Quality of Effluent from Marsh Treatment System Pilot Plant

1975	*BOD*[a]	*COD*[a]	*O-P*[a]	*t-P*[a]	*Cond.*[b]	*Turb.*[c]	*Colif.*[d]	*NO*$_3$[a]	*pH*
26 August	230.00	272.44	3.37	3.41	1540	83	5.158	.37	7.05
28 August	NS	NS	NS	NS	NS	NS	NS	NS	NS
4 September	NS	NS	NS	NS	NS	NS	NS	NS	NS
9 September	62	79.20			1050	44		0.28	7.05
16 September	70	72.5	16.90	16.97	1390	39	4.477	0.39	7.20
24 September	50	76.8	13.80	13.45	1400	45		0.18	7.10
30 September	40	49.92	12.18	11.83	1280	43	5.097	0.20	7.12
7 October	81	128.0	19.45	19.67	1400	34	5.491	0.22	7.3
15 October	72	105.6	18.32	19.67	1350	36	5.580		7.10
21 October	53	64.6	15.16	14.85	1400	34	5.301	0.19	7.78
28 October	65	119	16.34	15.47	1260	31.0	4.301	0.08	7.48
4 November	162	356.96	22.93	23.59	1650	37		0.18	7.30
Mean	72.8	116.95	16.76	16.68	1353	38.11	5.249	0.22	7.27
Standard Deviation	35.7	93.61	3.19	3.68	159	4.96	5.168	.094	0.23

[a] concentrations in mg l^{-1}
[b] micromhos
[c] J.T.U.
[d] log units
NS = No Sample

to decrease along the flow path from the influent to the effluent end of the marsh. They also tend to be higher in the stratum below the bulrush roots.

Other experiments indicated that the control basin without emergent vegetation performed almost as well as basins planted with bulrush in terms of removal of BOD$_5$, COD, total phosphorus, orthophosphate, and coliform bacteria. Decomposition is primarily due to microflora and fauna growing in the substrate as well as physicochemical processes.

The bulrushes are important primarily for aesthetic reasons and odor control, although they do increase treatment efficiency by several percent.

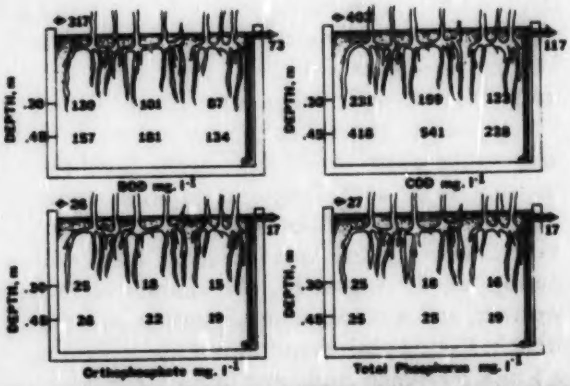

Fig. 2. Mean concentrations of BOD$_5$, COD, orthophosphate and total phosphorus, in influent, effluent and at 30 and 45 cm depth in artificial marsh pilot treatment plant receiving primary municipal sewage effluent.

It is important to design marsh treatment systems such that the water level is below the gravel surface. If standing water is not present, mosquitoes and similar insect pests cannot breed.

Wisconsin winters tend to be severe and our marsh systems were frozen from about December into March. During these periods the green parts of the bulrushes died. In spring new shoots grew from the rhizomes of the plant. Each season the number of bulrush shoots increased as the plants spread by growth of the rhizomes.

Since the artificial marshes are frozen during the winter, they cannot function as treatment systems. However, there are many situations where lack of treatment capacity during the winter is not critical. Indeed, in areas which do not experience killing frosts, research might reveal that marsh treatment systems could function all year.

POTENTIAL FOR SMALL-SCALE WASTE TREATMENT

Septic tank effluent has undergone mechanical settling and some anaerobic decomposition. It is chemically and bacteriologically similar to the primary sewage effluent used in our experiments. A small artificial marsh should yield results similar to our pilot plant (Table 1). Table 3 gives the chemical and bacteriological characteristics of the marsh effluent.

An artificial marsh suitable for a single-family summer home in Wisconsin would require a sloping trench lined with 20 mil plastic and filled with 15 cm of pea gravel. Initial studies indicate this should be 5 m wide and 30 m long although further research should be done to determine if the size could be reduced. The degree of slope would be determined by the hydraulic conductivity of the gravel. If the loading rate is 1.5 m^3/day (400 gal/day) and the conductivity is 41 m/day (1000 gal/day/ft^2), then the required slope would be 0.05. Bulrush rhizomes would be planted about 0.5 m on center. A sample design is shown in Figure 3.

A wet well would collect septic tank effluent and a sump pump would periodically flood the marsh with effluent. Evapotranspiration would reduce the volume of the wastewater. In our experiments this amounted to about 1.16 cm/day during July 17-August 21. This amount varies with weather, and is offset by precipitation into the trench. During peak evapotranspiration periods, a 5 m × 30 trench could lose up to 1.7 m^3/day. Under these conditions discharge from the marsh could fall to zero.

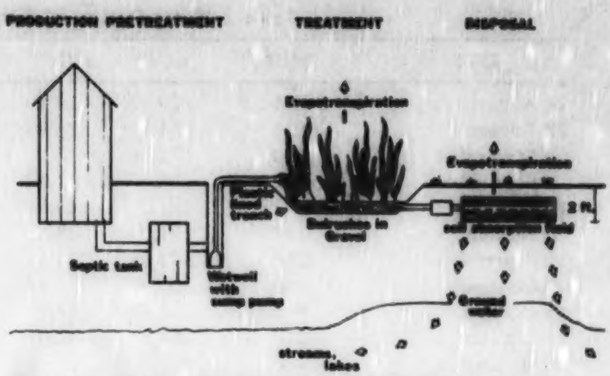

Fig. 3. Proposed design of artificial marsh treatment system for small-scale waste treatment.

Following marsh treatment the water could be discharged into the subsurface through conventional drain fields or cesspools with much less danger of ground-water contamination. In sandy soils part of the trench might be unlined and the effluent simply allowed to seep into the ground.

One of the significant ground-water pollutants is nitrate. The primary effluent we were treating was very low in nitrogen. While the marsh effluent had only 0.22 mg l^{-1} nitrate, it also had only 0.29 mg l^{-1} nitrate in the influent. Complete nitrogen analyses were not made and this would be a necessary area of future research.

Much more research is needed on small-scale marsh treatment systems to thoroughly evaluate their potential. The next step should be the construction and testing of some actual systems using septic wastes.

Marsh systems have the potential to provide an inexpensive individual treatment system which is virtually automatic and has few maintenance requirements. They could significantly reduce ground-water contamination from individual waste disposal systems.

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DISCUSSION

The following questions were answered by C. W. Fetter after delivering his talk entitled "Potential Replacement of Septic Tank Drain Fields by Artificial Marsh Wastewater Treatment Systems."

Q. *What were the typical input and output coliform densities per milliliter? Was the marsh effluent ever examined for bacterial pathogens, virus or stable organics?*
A. The average influent value was 7.314 log coliform units and the effluent had an average of 5.249 log coliform units. The effluent value is lower than the average coliform count of a stream which we studied in northeastern Wisconsin. The stream received a chlorinated secondary effluent and had an average of 4.76 log coliform units above the outfall and 5.54 log coliform units 100 meters below the outfall. Incidentally, following passage through 1800 meters of marsh, the coliform count in the stream then dropped to 4.68 log units.

We made no analyses of pathogenic bacteria, virus or stable organics. Some published reports indicate some species of *Scirpus* can degrade phenol (Kickuth, R., 1969).

Q. *What caused the increase in total dissolved solids?*
A. We don't know, but we noticed a similar increase in dissolved solids in a study of a natural marsh. This could be due to an increase in dissolved organic acids from the marsh plants.

Q. *Was the pilot plant operated under saturated conditions?*
A. Yes, water levels were kept at the gravel surface.

Q. *What is the life expectancy for a marsh treatment system for a single family dwelling? How long would the PVC liner last?*
A. The life expectancy of a marsh treatment system depends upon the suspended solids which accumulate in the gravel. We have no data on how long this process might take.

The PVC lining the basin was necessary for the experimental design since we didn't want unmetered water seeping from the experimental basin. In soil types which are not highly permeable, a basin liner would not be necessary. The bulrushes should not have the roots dry for extended periods, so the basins should be able to retain some water. A clay or silt lining would be better than a PVC lining for home systems since it would aid in phosphorus removal.

Q. *Are there other plants which could function like bulrushes? Could these be effective in arid climates?*
A. There are several species of bulrushes—*S. lacustris, S. validus, S. acutus* and *S. fluviatilis.* The cattail, *Typha,* functions in a similar manner, but is difficult to transplant. Dr. Käthe Seidel (1976) of the Max Planck Institute of Krefeld, West Germany, has experimented with a number of European plants. There has also been some work at the N.A.S.A. National Space Technology

Laboratories using water hyacinth (*Eichhornia crassipes*) and alligator weeds (*Alternanthera philoxeroides*) in biological treatment systems. If these plants are maintained with the roots in a saturated environment, they would grow in arid climates.

Q. *What is the impact of heavy metals on the system?*
A. We did not study heavy metal uptake. The wastewater source did not contain any industries which produce a heavy-metal-rich effluent. Other studies have shown that water hyacinths and alligator weeds do remove heavy metals and could be used in a heavy-metal treatment system if they are harvested (Wolverton, Barlow and McDonald, 1976, Chapter 17, in *Biological Control of Water Pollution*, University of Pennsylvania Press).

Q. *Could winter treatment be continued in northern climates if warm effluent is used?*
A. Not if there is a long retention time. A marsh system with standing water could be made much smaller, and enclosed in a greenhouse. This type of system would no doubt function, but we don't know the treatment efficiencies with lower temperatures and shorter daylight periods.

Q. *Could you recommend this system for institutions, such as schools, which have intermittent loads?*
A. Yes, if the bulrush roots are kept moist during periods when the system is not in operation.

Q. *Was there a control bulrush group which did not have effluent applied?*
A. Not as such, although my coworkers have done some studies of growth rates in natural, unpolluted marshes.

Q. *Does this system differ from a sewage lagoon treatment system?*
A. Yes, since sewage lagoons often have anaerobic areas with associated odor problems.

Ground-Water Pollution Aspects of Land Disposal of Sewage from Remote Recreation Areas[a]

by Nils Johnson[b] and Dean H. Urie[c]

ABSTRACT

Local ground-water pollution problems are often associated with disposal of sewage from remote recreational areas. Ground water in these areas is routinely used for domestic purposes without treatment. Sewage treatment facilities are often prohibitively expensive. Transporting sewage to municipal sewage treatment facilities is often equally costly, or adequate facilities are not available.

A soil incorporation method was tested at two field sites in Michigan's Upper Peninsula to evaluate its impact on ground-water quality. Liquefied campground sewage was injected at 15 cm (6 in.) depths using a liquid manure system. Average dosage levels were 2.7 metric tons per hectare (1.2 ton/acre) of dry sewage solids. The application rate was equivalent to a fertilization rate of 116 kg/ha (104 lb/a) of total nitrogen. Field tests were conducted on adjacent strips of Kalkaska sand soils in 1973, 1974, and 1975.

Changes in ground-water quality in the zone immediately beneath the water table were evaluated by analyzing samples from randomly located wells. Test wells were located directly beneath the treated area and up to 30.5 m (100 ft.) away in the direction of ground-water flow. Nitrate levels in ground water at 3.6 m (12 ft.) depths were higher than control levels from the treated zone to the limit of the test well sampling. Levels of nitrate did not exceed the limits for potable water (10 mg/l NO_3-N). No fecal coliform organisms were detected in ground-water samples. Laboratory tests of the filtration capability of the A and B horizons indicated a high level of filtration of polio virus.

Fertilization with sewage increased the biomass of native herbaceous vegetation by 410 percent. Nitrogen content of the treated vegetation was 63 percent above controls. Approximately 48 percent of the added nitrogen fertilizer was incorporated in herbaceous foliage at the end of the first growing season.

Site selection guidelines for use by National Forests include remote location of incorporation sites, no potable-water sources within one-half mile downgradient from the site, infiltration rates between 5 and 25 inches per hour, slopes less than 5 percent, and dosage rates which will not exceed 50 kg/ha (56 lb/a) of mineralized nitrogen per year. Injection depths should provide complete soil coverage but sewage should be placed above the B horizon for maximum use of nutrients by plants.

The soil incorporation method has been approved for selected sites by the Michigan Department of Natural Resources and the U.S. Environmental Protection Agency, with ground-water monitoring required at each site.

INTRODUCTION

The simplest and oldest way to dispose of sanitary wastes is to drop them on the soil and let nature take its course. This fact is derived from the most casual observations of animal and primitive man. When modern civilizations result in concentration of wastes, the mechanics become much more complex and the environmental hazards more serious.

Disposal of human waste on the Hiawatha National Forest in northern Michigan required a method that was inexpensive and yet gave satisfactory renovation. The volume of waste from campground vault toilets located throughout the Forest is approximately 15 to 20 thousand gallons per year.

Vault toilets were installed on the Hiawatha

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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National Forest because of the extremely sandy soils located in the area. These soils have subsoils with very rapid percolation rates and little capacity to renovate wastes. To protect ground water and lake quality, sealed vaults were installed on the Forest. These vault toilets on the Hiawatha National Forest are sealed concrete tanks, having a volume of approximately 500 to 1,000 gallons. A building containing a toilet is placed on top of the vault. This dry system requires periodic pumping with a standard septic tank pumper to empty the vaults.

Several alternative methods were available to dispose of the vault sewage. The wastes could be spread on the ground surface where biodegradation would occur. This would leave human waste exposed however. U.S. government land administered by the U.S. Forest Service is open to public access. For public health reasons, this alternative was rejected.

The waste could be taken to municipal waste treatment plants located adjacent to the Forest. This alternative was rejected for two reasons. First, the plants do not have holding lagoons, and a single dose application of concentrated waste would upset the biological operation of the plants. Second, this method of disposal would require expensive, long hauls because of the remote locations of campgrounds.

The third alternative was to build a wastewater treatment facility to dispose of toilet wastes. This is a viable alternative; however, initial investment is high and the system would require an operator and continuing maintenance.

GENERAL DESCRIPTION OF THE AREA

The two incorporation sites are located in Alger and Delta Counties of Michigan's Upper Peninsula. The aquifers affected by the tests are sandy drift deposits of Wisconsin Age that are approximately 15 m (50 ft.) thick overlying limestone of Ordovician Age. Ground-water levels are between 3-5 m (10-16 ft.) below the ground surface. Soils in the test areas are sandy, mixed, frigid, Typic Haplorthods and have been used for agriculture. Soil cation exchange capacities are about 5-10 m.e./100 g. The average rainfall is 813 mm (32 in.) and is evenly distributed throughout the year. The climate is continental with a moderating lake effect.

Many ground-water wells in the area are shallow sand points and thus protection of the quality of ground water in the shallow aquifer is of paramount importance.

METHODS

A system referred to as soil incorporation was initiated for waste disposal. Wastes were injected at a depth of about 15 cm (6 in.) below the ground surface. Renovation of the waste is by micro-organisms present in the soil, filtration and adsorption by soil materials above the ground-water table, and nutrient uptake by the vegetation present on the incorporation site.

A series of test wells consisting of 3.18 cm (1.25 in.) 80 mesh well screens were placed just below the water table. Porous ceramic samplers were also placed in the upper three feet of the saturated zone. These wells and samplers were periodically sampled for analysis of ground-water quality. All water samples were preserved and analyzed according to *Standard Methods* (APHA, 1971).

Soil samples were collected by the use of a punch auger. Control and treatment samples were composited and analyzed by micro-kjeldahl and emission spectrolysis.

Vegetation for analysis was collected using a 929 cm^2 (1 ft.^{2}) sampler. Samples were dried and ground for analysis. Analysis of total N was by micro-kjeldahl. Cations were analyzed by emission spectrolysis.

Ground-water flow direction was determined by the construction of a flow net using four observation wells. Wells for analysis of ground-water quality were located at incremental distances in the direction of ground-water flow. Lateral locations were randomly selected within the interior 50 percent of the zone of expected fertilizer effect.

KENTUCKY CCC SITE–CRITICAL TEST

A preliminary test began in 1972 on the Munising Ranger District of the Hiawatha National Forest in northern Michigan. The site presented a relatively high risk of ground-water contamination because of (1) highly permeable sand soil of the Kalkaska series, and (2) water tables at only 1.8-2.4 m (6-8 ft.) depths.

A 0.08 ha (0.2 acre) plot was injected with 5,300 l (1,400 gal.) of campground vault wastes in August 1972, at the rate of 6.2 l/m (0.5 gal/ft.) of furrow. Furrows were spaced 0.69 m (2.25 ft.) apart. The sewage was injected at a 15 cm (6 in.) depth using a Clay Honeywagon tank trailer equipped with two injector shoes which opened two furrows, injected the sewage and closed the furrows in one operation. (Mention of product names does not indicate endorsement by the U.S.

Table 1. Composition and Application Rates of Sewage Vault Wastes Injected at 65,500 l/ha at Kentucky CCC

	Concentration	*Dosage Rate*
Total solids	2.86%	1869 kg/ha (1668 lb/a)
Total N	13.17% (dry wt.)	224 kg/ha (200 lb/a)
Total P	2.0% (dry wt.)	34 kg/ha (30 lb/a)

Forest Service.) Composition and dosage rates of the vault sewage are shown in Table 1.

RESULTS OF NUTRIENT STUDY

Monitoring wells were installed (Figure 1). After two years of monitoring, no elevated nitrogen levels have been found at either FC-2 or W-3 located 6.1 m and 12.2 m (20 and 40 ft.) down the ground-water gradient, respectively. No significant phosphorus increases have been detected in any ground-water samples.

High nitrogen values are found only directly under the plot and adjacent to the plot. Increased nitrogen values have not been detected in wells located further down the ground-water gradient.

Nitrate nitrogen rose to a peak of about 50 mg/l in wells directly under the treated plot following snow melt in 1973, but dropped sharply within six months. Nitrate and nitrite nitrogen accounted for practically all the total nitrogen as would be expected under conditions of nitrification. The increase in nitrogen levels following treatment is attributed to the fact that the (1) water table is only 1.8-2.4 m (6-8 ft.) below the surface, (2) Kalkaska sand soil has a rapid infiltration rate, and (3) nitrogen content analysis of the waste was 1.5 times higher than average

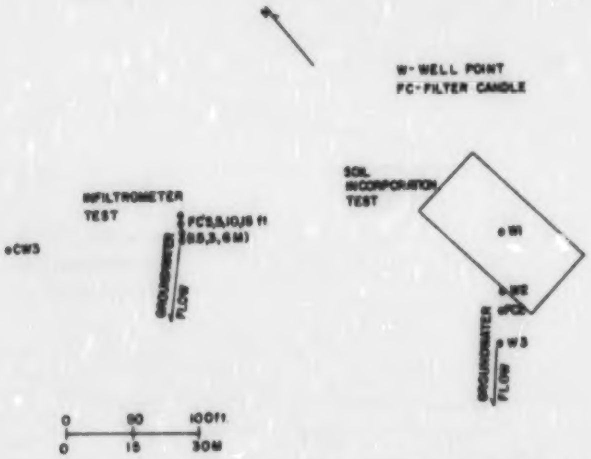

Fig. 1. Location of sewage treatment tests and ground-water quality sampling points, Kentucky CCC test.

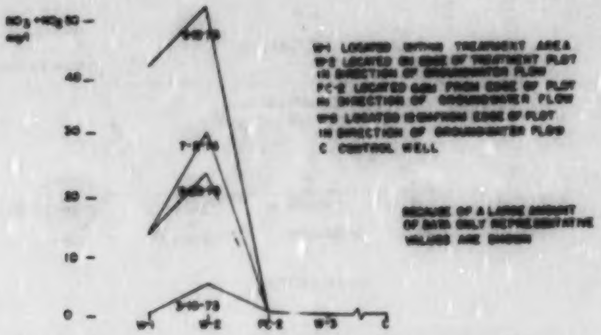

Fig. 2. Nitrite plus nitrate nitrogen levels in ground water following sewage injection at Kentucky CCC Site.

values measured in subsequent treatments. Much of the horizon development of the soil has been disturbed during occupancy of the site by a Civilian Conservation Corps Camp during the 1930's. The relatively high nitrate levels could be attributed to poorer soil-nutrient relationships resulting from this disturbance.

BACTERIOLOGICAL RESULTS

Through a cooperative study with Michigan Technological University, by Tluczek (1974), the movement of the fecal bacterial indicators was monitored.

The raw waste was characterized as follows:

5.52×10^7 Total Coliforms (TC)/100 ml
1.74×10^7 Fecal Coliforms (FC)/100 ml
3.28×10^7 Enterococci (*Streptococcus facalis)* (EC)/100 ml

A circular 2.52 square meter (25 ft.^{2}) single ring infiltrometer was placed over two injection furrows and irrigated with 161 mm (6.35 in.) of tap water at the rate of 380 mm (15 in.) per hour.

Under the artificial rainfall condition, total and fecal coliform and enterococci organisms were detectable at 0, 1.5, 3.1, 4.6 m (0, 5, 10, and 15 ft.) down the ground-water gradient directly after treatment. Enterococci dropped to zero one day after injection and remained at zero at the two most distant wells. Enterococci were detected at low levels for one month after treatment at 0 and 1.5 m (0 and 5 ft.). No enterococci were detected in any ground-water samples after this date.

Fecal coliforms were detected at 0, 1.5, 3.1, and 4.6 m (0, 5, 10, and 15 ft.) down the ground-water gradient directly after treatment (August 1972) through the first week of June 1973. Counts ranged from approximately 1 to 9 organisms per 100 milliliters. Highest counts occurred directly under and adjacent to the plot (0-1.5 m).

Total coliform organisms were detected at

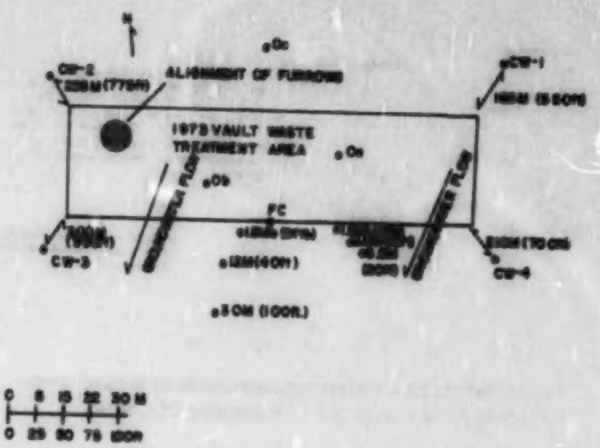

Fig. 3. Sewage treatment test area and location of ground-water quality sampling points, Birch Farm test.

all treatment wells directly after treatment on August 17, 1972, until the middle of December 1972.

Elution and localization studies of reconstructed soil columns showed that bacteria were mechanically filtered from the effluent by the somewhat finer-textured soil horizons (A & B horizon).

Longevity of fecal coliform and enterococci organisms in the soil at the Kentucky CCC Site showed that in approximately one year, fecal coliforms were down to 42 organisms per gram of soil and enterococci were zero.

BIRCH FARM PILOT TEST

During the summer of 1973, a project sized test using vault-toilet waste from three ranger districts was begun.

The procedure was similar to that used at the Kentucky CCC Site. A large open field, Birch Farm, occupied by soil of the Kalkaska sand series with the water table at approximately 3.6 m (12 ft.) below the surface was used. The site received approximately 53,000 l (14,000 gal.) of waste on a 0.81 ha (2 acre) tract during June through September 1973, and approximately 18,900 l (5,000 gal.) on 0.4 ha (1 acre) during June through September of 1974, and a similar amount in the summer of 1975. Monitoring of the site was accomplished by the design shown in Figure 3.

RESULTS OF BIRCH FARM NUTRIENTS STUDY

Dosage Rates

Analysis of the sewage has shown a high variability in the total solids content from 3.2 to 0.4 percent. This variability is due to the amount of water added to pick up the sewage from the sewage vaults, and the presence or absence of ground-water seepage among the various campground locations. Average concentrations and nutrient dosage rates are shown in Table 2.

Table 2. Composition and Dosage Rate of Vault Wastes Injected at 65,000 l/ha at Birch Farm

	Concentration	Dosage Rate lbs/acre	Dosage Rate (kg/ha)
Total solids	2.02%	1,173	1,314
Total N	8.83% (dry wt.)	104	117
NH^3-N	7.63% (dry wt.)	90	101
Total P	1.37% (dry wt.)	16	18
K	7.7% (dry wt.)	90	101
Ca	2.6% (dry wt.)	31	35
Mg	0.1% (dry wt.)	1	1

Total nitrogen and nitrite plus nitrate as nitrogen rose slightly directly under and adjacent to the plot. There have been no increases detected at 40 feet from the edge of the plot down the ground-water gradient (Figure 4). Although nitrate plus nitrite levels rose to a maximum of

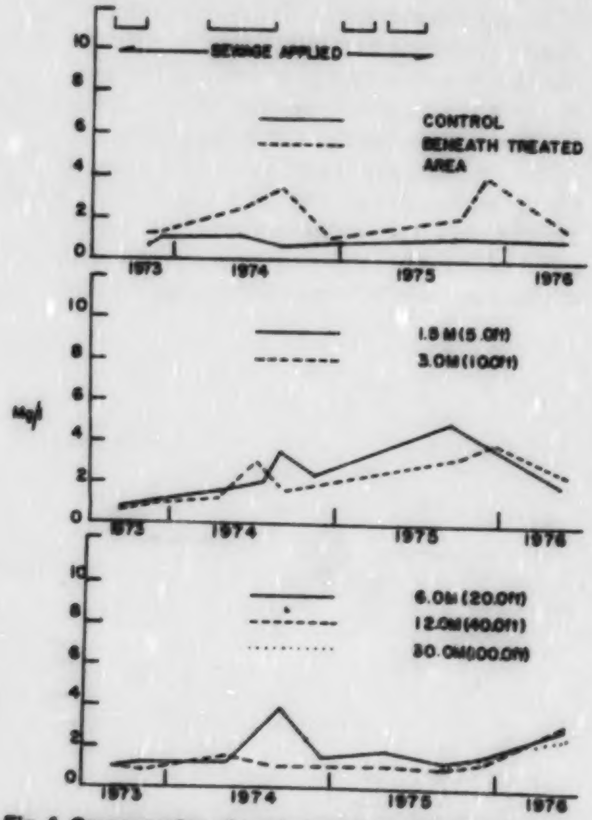

Fig. 4. Concentration of total nitrogen in ground-water samples removed from wells at Birch Farm test site.

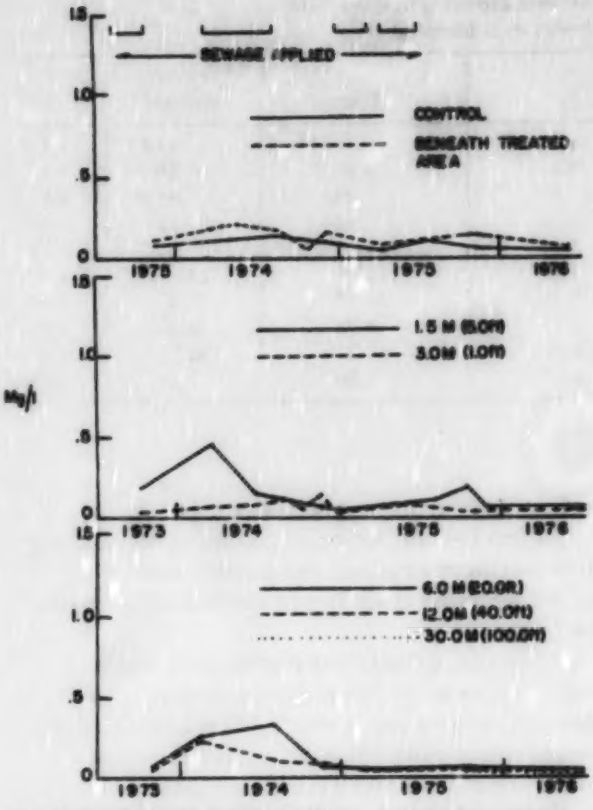

Fig. 5. Total dissolved phosphorus in ground-water samples removed from wells at Birch Farm test site.

8.9 mg/l as nitrogen, they did not reach the U.S. Public Health Service Standard of 10 mg/l. Total dissolved phosphorus as PO_4 has remained near background levels as would be expected on sand soils with a high ability to absorb phosphorus (Figure 5).

BIRCH FARM BACTERIOLOGICAL RESULTS

The waste contained the following bacterial population ranges:

$1.0 \times 10^5 - 2.4 \times 10^6$	Total Coliforms (TC)/100 ml
$1.08 \times 10^4 - 2.6 \times 10^4$	Fecal Coliforms (FC)/100 ml
$2.8 \times 10^5 - 4.0 \times 10^5$	Enterococci *(St. facalis)* (EC)/100 ml

Measurement of fecal and enterococci organisms indicated that no enteric organisms reached the ground water at any of the observation wells.

The absence of bacteria in ground water at the Birch Farm Site versus positive findings at the Kentucky CCC Site can be attributed to a number of variations in the sites and experimental technique.

The water table at Birch Farm compared to the Kentucky CCC Site is 3.0 m and 1.8 m (10 ft. and 6 ft.) respectively. This provided a thicker filter for bacteria removal at Birch Farm. The soil at Birch Farm has more fine-textured particles and a higher organic matter content allowing for more bacteria to be mechanically filtered in the upper horizons of the soil.

Comparing the two sites, the Kentucky CCC Site was artificially irrigated and the Birch Farm Site was not. Thus more downward movement of water allowed bacteria to reach the ground-water table at the Kentucky CCC Site. The Kentucky CCC Site had much soil disturbance in the past; the Birch Farm Site has not been disturbed. Soil-fauna-nutrient relations are thus much better at the Birch Farm Site.

VIROLOGY STUDY

Laboratory studies of virus-innoculated 48 cm soil columns showed that soil columns flushed with one large volume of water were able to retain greater than 99.99% of the T_4 bacteriophage and 99.9% of the poliovirus.

The study concluded that there is only a low-grade danger of virus contamination of ground water. The studies were conducted in reconstructed soil columns only 48 cm in length. Under actual site conditions where there is approximately 3 meters of soil above the water table, contamination results would be expected to be proportionally lower than in laboratory studies.

VEGETATIVE RESPONSE

The operation of the system is highly dependent on utilization of the nutrients by the vegetation that is present on the site. Table 3 shows the levels of foliar elements of the treatment versus the control plots in August 1975.

Total nitrogen concentrations in plant tissues increased significantly following fertilization with vault wastes:

	Birch Farm		Kentucky CCC	
	Treated	Control	Treated	Control
July 1974	1.88%	1.46%	1.92%	0.98%
August 1974	1.24%	0.76%	1.06%	0.94%

The volume of dry matter on the site in above-ground herbaceous vegetation increased an average of 410 percent on injected areas. During July 1974, approximately 48 percent of the nitrogen added as vault wastes was incorporated in the increased biomass.

SOIL FERTILITY EFFECTS

The Kentucky test site with its disturbed topsoil condition presented extremely variable

Table 3. Concentration of Nutrient and Mineral Elements in Plant Tissue on Soil Incorporation Plots (Mean of 5 Samples)

	Kentucky CCC			Birch Farm		
	Treated	*Control*	"t"	*Treated*	*Control*	"t"
Phosphorus %	.184**	.136	6.16	.177**	.118	4.3
Potassium %	1.48**	.72	9.8	1.65**	.80	4.4
Calcium %	2.11	.221	.37	.388	.453	0.8
Magnesium %	.098*	.076	2.85	.128	.115	.89
Aluminum ppm	33*	134	2.45	119	406	1.75
Iron ppm	109*	221	2.34	162**	321	4.2
Zinc ppm	44.7*	52.6	2.38	55.1*	74.8	2.83
Copper ppm	6.44	9.18	1.31	4.96	8.8	1.5
Manganese ppm	152.5	270.1	1.95	61.4	198.3	1.15
Boron ppm	4.22*	6.24	2.67	6.80	9.34	1.08

Significance — * 5% level
** 1% level

materials for analysis. The cation exchange capacity was also very low (Table 4). Injection of vault wastes generally resulted in increased total soil nitrogen values. Potassium values were consistently higher and bases were reduced, probably due to slightly lower pH values and displacement by ammonia from the injected sewage.

DISCUSSION

The soil incorporation of vault toilet wastes has proved to be a reliable, low-cost method for the disposal of concentrated wastes from recreation areas on the Hiawatha National Forest. The annual cost of disposing of the 57,000 l (15,000 gal.) of wastes from the three ranger districts is approximately $2,000 per year.

Bacteria and nutrient results at the Kentucky CCC Site indicated that bacterial and nutrient contamination of the ground water can be expected under extreme conditions. These conditions are (1) poor soil development due to past disturbances resulting in less than optimum uptake of available nutrients from the waste, (2) relatively shallow water tables, (3) high soil flushing rates with water after incorporation of wastes.

Virus contamination of ground water resulting from soil incorporation is a relatively remote hazard when the guidelines in the following section are followed.

Bacteria and nutrient studies from a pilot study conducted at Birch Farm gave results that indicated when a site is selected that has (1) a deep ground-water table, (2) good soil development, and (3) operates under normal precipitation conditions, the water quality of the ground water will not be appreciably affected.

Good soil-nutrient relationships are a must if the site is to renovate the waste and the vegetation is to utilize the resulting available forms of nitrogen. An indication of the nutrients that are made available from the incorporation of waste, is the great increase in vegetation growth that has been observed following incorporation of waste at a site.

The system depends on filtration by the soil material above the ground-water table and nutrients from the waste being used by existing or planted vegetation. With these thoughts in mind and the results of the studies on the Hiawatha National

Table 4. Soil Nutrient Analysis in Sewage Injection and Control Sites — Kg/ha

	Total N		Total P		Potassium		Calcium		Magnesium		C.E.C.		pH	
	Kg/ha										meq./100 g.			
Date	*Treat*	*Cont*	*Treat*	*Cont*	*Treat*	*Cont*	*Treat*	*Cont*	*Treat*	*Cont*	*Treat*	*Cont*	*Treat*	*Cont*
Kentucky CCC:														
8/74	835	610	78	72	54	27	307	460	25	37			5.6	5.9
4/75	860	1230	43	58	27	14	156	460	12	52			5.4	5.9
10/75	990										3.3		5.2	
Birch Farm:														
8/74	1900	1560	47	81	81	54	461	615	25	12			5.9	5.9
4/75	1820	1790	39	44	92	49	435	409	38	38			5.6	5.6
10/75	2170	1635									14.4	16.2		

Forest, the following *Guidelines for Use* are proposed to guide future and existing soil incorporation projects.

GUIDELINES FOR USE

1. Injection depth should be between 10 to 20 cm (4 to 8 inches). This depth is required to take advantage of favorable chemical and bacteriological relationships in the surface soil horizons necessary for renovation of vault wastes, and to provide covering of the waste.

2. The site is to be located at least 180 m (200 yd.) from any residence or regular travel lane. This is the distance from residences required for disposal of wastes from septic tanks in Michigan. This is a minimum and in actuality should be a minimum of a half mile. The ability of this system to be used in remote areas is an advantage over other systems.

3. Application rate is not to exceed 6.2 l (1.6 gal.) of waste per meter (3.3 ft.) of furrow at 0.69 m (2.25 ft.) spacing. This was the rate and spacing used on the Hiawatha National Forest and proved to be a workable dosage. These values could possibly vary in other areas based on equipment used and soil capabilities.

4. Site should be nearly level with no drainage ways within the area or immediately adjacent to the injection area. General topography should have a maximum slope of 5.0% and should not have erratic undulations. This is so overland flow, if present, will not carry contaminants into surface waterways.

5. Ground-water levels must be at least 3.0 m (10 ft.) below the existing ground level. Experiments on the Hiawatha National Forest have shown this to be the most acceptable depth for waste renovation.

6. The area used should never be subject to flooding and have bedrock at least 3 m (10 ft.) below the surface.

7. The site should have less than 10% stone over 2 cm (5 in.) diameter in the surface foot. This is a requirement based on equipment limitations and could be changed if stronger equipment were available.

8. Infiltration rates are important when considering the need for the liquid faction of the waste to move into the soil without oozing occurring at the surface. Profiles should be homogeneous without restrictive pans or banding within the surface 3 m (10 ft.) and have infiltration rates of 12.7 cm to 63.5 cm (5 to 25 in.) per hour.

9. Soil texture is also an important consideration when rating soils due to the fact that finer soils are better suited for filtration of bacteria and virus and for the renovation of chemical constituents of vault waste. A listing of soil textures that fulfill the requirements for renovation and infiltration in decreasing order of preference are:
   a. Well-drained sandy loams
   b. Well-drained fine sandy loams
   c. Well-drained loamy sands
   d. Well-drained sands
   e. Well-drained loams

10. Ground water shall be monitored at all sites for the following chemical and biological parameters:
    a. Chemical parameters:
       1. Total nitrogen
       2. Nitrate and nitrite – N
       3. Total dissolved phosphorus
       4. Ammonia nitrogen
       5. Specific conductance
    b. Biological parameters:
       1. Total coliform organisms
       2. Fecal coliform organisms

11. The site must be well vegetated with grasses and forbs or be vegetated through seeding to effectively utilize and recycle nutrients imparted to the soil by the injected vault waste.

12. The area shall be fenced and posted to exclude entrance by persons who happen upon the site.

13. Loading rates of waste should not exceed 50 kg/ha (56 lb/a) of mineralized nitrogen per year to insure protection from nitrate buildup in the ground water.

REFERENCES

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Buller, Richard S. 1974. Potential viral hazards of land disposal of human wastes. Unpublished M.S. thesis–Michigan Technological University, Houghton, Michigan.

Tluczek, Louis J. M. 1974. Public health considerations of the soil incorporation of campground sanitary wastes. Unpublished M.S. thesis–Michigan Technological University, Houghton, Michigan.

DISCUSSION

The following questions were answered by Nils Johnson after delivering his talk entitled "Ground-Water Pollution Aspects of Land Disposal of Sewage from Remote Recreation Areas."

Q. *Do you think that the reduction of nitrates with time is caused by increased plant growth?*
A. Yes. At the Kentucky Site it is because of vegetative use of the nitrogen and the nitrogen source (waste) being lost, partially by leaching.

Q. *Did you determine TDS in ground water?*
A. Indirectly, by the use of specific conductance and then using a correction factor to estimate TDS. A periodic evaluation of cations and anions showed that nitrogen was the major added substance in the ground water.

Q. by Joe Dix. *Did you harvest the vegetation?*
A. No, because we did not have a use for it. Future applications should consider this as it would serve as a method of nutrient removal from the site and as a source of useful animal fodder.

Q. *In terms of human activity, what happens to the site when abandoned?*
A. The pilot area is dedicated to wildlife management and will be allowed to revert to this use.

Q. *What is the natural rainfall-longevity intensity at the test sites?*
A. The 100-year, 24-hour storm is 6.35 inches,
the 50-year, 24-hour storm is 4.25 inches,
the 10-year, 24-hour storm is 3.25 inches,
the 5-year, 24-hour storm is 2.90 inches,
the 2-year, 24-hour storm is 2.25 inches,
the 1-year, 24-hour storm is 1.95 inches.

Q. by Martin J. Allen. *Did you examine whether your injection practices resulted in increased bacterial densities (non-coliforms) within the ground water since excessive bacterial population can desensitize the bacteriological methods used to isolate total or fecal coliforms?*
A. Total plate counts were not performed; however, selective mediums and the MPN method were used which tend to decrease this problem. Wherever the MF technique was used as a check later in the study, overgrowths, which are often prevalent when high concentrations of non-coliform organisms are present, were not seen on the plates.

Q. by Mike Apgar. *How much confidence do you have that samples downgradient from the infiltrometer site were really intercepting contaminant flow? Was bypassing of the sampling points possible?*
A. Wells were approximately 5 feet or deeper below the water table. Samples were always taken during or directly after recharge period (i.e. spring, fall) and periodically at other times. Therefore, we are quite confident of our sampling. It is possible that nutrients could have passed the well during major rainfall events but they would have been of lesser concentrations than those reported.

Q. by Jim Kimball. *What did you use for odor control in the holding tank?*
A. Nothing. The vault toilets have a draft ventilation system.

Q. *Did beer cans and other trash often disposed of in vaults plug the pumper truck?*
A. No. The septic tank pumper did not usually pick up this material. Whenever foreign material plugged the system, the positive pressure created by the injector was sufficient to clear the lines.

Q. by William A. Trippet. *Are the inorganic (NO_3-PO_4) peakings over the years of data collection due to rainfall?*
A. No. Peaks are usually related to recharge periods (i.e. snow melt, vegetative dormancy, etc.). Controls do not show the amplitude in peaks that the treated area does, thus indicating that peaks are related to the treatment.

Q. by Charles P. Vanderlyn. *Have you investigated alternative waste treatment systems; for instance, chemical units, biological aeration systems, or incineration of sanitary wastes?*
A. Yes. We use these systems in some instances. We offer this method as one that is compatible with the land resource, builds soil properties, does not create an end product that must be disposed of, and is environmentally sound. It is only applicable with dry human waste systems.

Q. by K. Childs. *What do you do with your sewage during the winter months? Don't your site requirements exceed those presently required by the State of Michigan for spray irrigation systems?*
A. Our recreation areas are not open during the winter months and, thus, no waste is generated during this time. We use this system between April and October.

This system is not a high leaching system nor is the disposal material a treated effluent, as with a spray irrigation system, so these requirements are not applicable.

Q. by Clinton Whitmer. *How long before the area may be impacted a second time?*
A. At the Birch Farm Site we reinjected the 1972 site after three years. We felt the land system could withstand reapplication because of the increased vegetative cover resulting from the initial application.

The Selection and Management of Feedlot Sites and Land Disposal of Animal Waste in Boise Valley, Idaho[a]

by L. L. Mink[b], C. M. Gilmour[c], S. M. Beck[c],
J. H. Milligan[c] and R. L. Braun[d]

ABSTRACT

Environmental effects of feedlot location and related land disposal operations can be minimized if proper knowledge of hazardous conditions are known and appropriate protective steps taken. Common guidelines often do not apply because of differing physical characteristics of local areas.

Analyses of various parameters within a soil profile beneath a feedlot revealed none of the chemical constituents present in high concentrations below the 23-foot depth. In areas where shallow ground water was less than 5 feet from the surface, the ground water was found to be affected by the feedlot. Other hazardous areas in feedlot location are flood-prone areas, areas of surface bedrock, and areas of excessive slope.

For land disposal operations, loading rates and frequency of application of feedlot waste should be adjusted in accordance with soil permeability, depth to ground water, and irrigation practices to minimize detrimental effects on ground-water quality.

INTRODUCTION

Early in 1970 the question of point source pollutants with respect to cattle feedlot locations and the disposal of feedlot waste became a major issue in Idaho. Of prime concern was the Boise River Valley located in southwest Idaho (Figure 1). It has been estimated that within the Boise Valley there are over 350,000 head of beef cattle being finished annually. Because of economic and environmental factors, feedlot waste is now viewed as a beneficial product rather than a waste to be disposed of at a minimum of effort and expense. Within the Boise Valley the practice of applying manure to crops is gaining acceptance because of the nutrient value and soil building characteristics of the manure.

Within the Boise Valley a major portion of the surface waters are diverted and used for irrigation. The irrigation return flow back to the surface-water system and the infiltration of irrigation waters into the shallow ground-water system presents a potential problem of surface-water and ground-water contamination due to feedlot location and/or management. From this analysis of the problem, the U.S. Army Corps of Engineers contracted with the University of Idaho and the Idaho Bureau of Mines and Geology in cooperation with local governing agencies to undertake a two-year study of feedlot locations and associated land disposal of feedlot wastes.

LOCATION

Boise Valley is a broad alluvial valley which lies at elevations ranging from 3000 feet in the upper (SE) portion of the valley to 2400 feet in the lower (NW) portion of the valley. Several low river terraces occur on both the north and south sides of the valley but are more pronounced to the south.

The valley is bounded on the north by what is termed the Boise Ridge and the south by a rolling upland plain. The mountains comprising the Boise Ridge area rise to elevations of over 6000 feet and are composed of granitic rocks of the Idaho batholith. Quaternary-tertiary sediments form a low foothills area between the granites of the Boise Ridge and the Boise River Valley.

The Boise River is the major surface-water system of the area draining several tributary valleys from the north. Much of the water from the Boise River is diverted and used for irrigation within lowlands of the valley. This diversion water and resulting irrigation have raised the water table over large areas of the valley to within a few feet of the surface.

The climate of Boise Valley is arid to semiarid. Summer months are dry and warm with cool

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[a] Supported by U.S. Army Corps of Engineers, Walla Walla District. Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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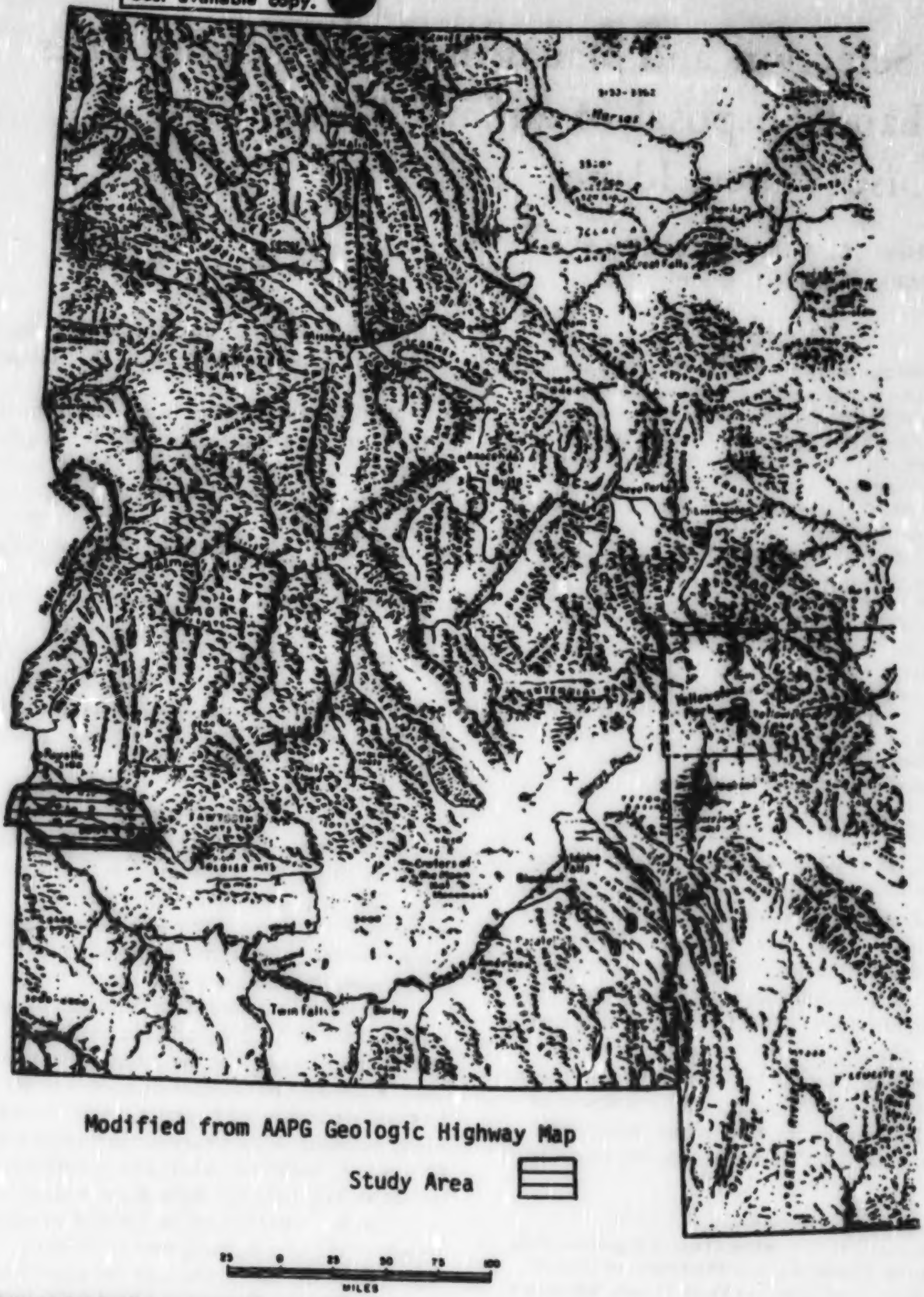


Modified from AAPG Geologic Highway Map

Fig. 1. Boise Valley feedlot and land disposal study area.

nights while winter months are generally moist and cold. Average summer daily maximum temperature is 84° F (29° C) and average winter temperature is 30° F (−1° C) for the valley. A majority of the precipitation occurs during winter months with average annual precipitation averaging 10-15 inches annually at the lower elevations.

ANALYSIS

Feedlot waste is generally considered an agronomic resource material, containing substantial quantities of nitrogen, phosphorus, potassium and other elements well established as valuable fertilizer material. It also is beneficial in increasing water retention and increasing organic matter content within soils. In all, the estimated value of feedlot waste if applied to crops is $3.36/ton (Table 1).

Several constituents of feedlot waste may become hazardous if proper disposal procedures are not followed. Often the nitrate nitrogen component and the coliform bacteria found in feedlot waste find their way into ground water or surface waters and create hazardous situations.

Feedlot Characteristics

Chemical composition of feedlot waste used in land disposal operations near Caldwell is summarized in Table 2. Hrubant *et al.* (1972) reported assorted species of entero-bacteria to number from 4.4 to 6.8 × 10^7 per gram of dry feedlot waste. To determine the effect of feedlot waste on the subsurface environment, two feedlots within the Boise Valley were studied in detail. Feedlot A was located in the Boise River Flood-plain where ground-water levels are less than 5 feet from surface. Feedlot B was located on an alluvial terrace where depth to ground water was over 80 feet.

Density measurements within Feedlot A found average dry densities of the feedlot surface to be 52.1 lb/ft^3 (0.84 g/cm^3) and average densities of the soil immediately beneath the manure pack to be 120.6 lb/ft^3 (1.93 g/cm^3). Volatile solids content of the manure on a dry-weight basis was 46.4% while the volatile solids of the soil was only 2.8%. The high density of the soil indicates compaction of a feedlot surface is not limited only to the manure layer, but extends into the upper portions of the soil horizon. It also indicates one may expect lower permeabilities and infiltration rates in the soil due to the lower porosity resulting from compaction.

Infiltration tests were conducted using both ring infiltrometers and rainfall simulator tests on both feedlots during the summer and fall of 1974. In all short-term tests (less than 6 hours), infiltration rates decreased with time resulting in a final intake rate of less than 0.08 in/hr. In the long-term infiltration tests (7 days), final infiltration rates averaged 0.0033 in/hr with all rates below 0.005 in/hr. At the termination of the 7-day tests the infiltration rings were removed and the manure and soil directly beneath the rings examined for moisture content. The manure was saturated but the underlying soil had a moisture content averaging 14.6%. This moisture content was actually less than the moisture content of surrounding soil which averaged 16.4%. This indicates vertical percolation beneath the manure packs will be under unsaturated flow conditions.

Table 1. Potential Value of Applied Beef Feedlot Wastes

Fertilizer Benefit	*Content/ Ton Manure*	*Price/ Pound*	*Value/ Ton*
Nitrogen/ton	4 lbs. N	$.325	$1.300
Phosphate/ton	4 lbs. P_2O_5	.256	1.024
Potash/ton	9 lbs. K_2O	.087	0.783
Fertilizer Value Per Ton Beef Feedlot Manure			$3.107
Physical Benefit			
Increased Water Retention Rate/Ton			$ .007
Increased Organic Matter Content/Ton			.250
Physical Value Per Ton Beef Feedlot Manure			$0.257
TOTAL POTENTIAL VALUE OF APPLIED BEEF FEEDLOT MANURE/TON			$3.364

Simulated rainfall tests were conducted to sample all feedlot surface conditions and to determine the effect of slope on surface runoff. Although average water application rates as low as 0.06 in/hr were maintained, runoff occurred. This is consistent with the low rates found utilizing the ring infiltrometers. Another purpose of rainfall simulation was to calculate the total amount of precipitation absorbed or abstracted by the manure pack. For initial dry conditions the volume of water retained by the manure pack was 0.56 inch and for initially wet conditions the amount averaged 0.23 inch. On two occasions simulated rainfall was repeatedly applied for two days. Total water-holding capacity of the manure was

Table 2. Composition of Caldwell Beef Cattle Feedlot Wastes (from Sikora, 1973)

Constituent	*Concentration*
Organic matter	51.06%
Organic carbon	29.69%
Organic nitrogen	0.77%
Soluble calcium, Ca^{++}	130 ppm
Soluble magnesium, Mg^{++}	133 ppm
Soluble sodium, Na^+	3,990 ppm
Soluble potassium, K^+	774 ppm
Soluble chloride, Cl^-	22,800 ppm
Soluble ammonium nitrogen, NH_4^+-N	202 ppm
Soluble nitrate nitrogen, NO_3^--N	137 ppm
Soluble nitrite nitrogen, NO_2^--N	2.8 ppm

estimated by summing the cumulative rainfall and allowing for evaporation and runoff. The average water-holding capacity of the manure pack was 1.33 inches which calculated to be 0.44 inch of water per inch of manure thickness. The moisture penetrated approximately 0.4 inch for a single application on a dense manure pack. Where water was applied repetitively on a dense manure pack, moisture penetration was no greater than 4 inches. For single application on a poorly formed manure pack over sandy soil, a penetration depth was found to be 3 inches or less.

Slope up to 12% had little apparent effect on infiltration rates and water-holding capacity of the manure pack, but slopes over 12% caused substantial reduction in water-holding capacity and increased surface runoff. This was found to be consistent with several other reported investigations.

Ground-Water Quality

Ground-water samples beneath three established feedlot pens and a nearby pasture were collected and analyzed for the following: pH, electrical conductivity, sodium absorption ratio, calcium, sodium, chloride, bicarbonate, orthophosphate, nitrite, nitrate, ammonium, chemical oxygen demand, fecal coliform, sulfate, potassium and carbonate.

General trends were found to exist among several of the parameters. Calcium, bicarbonate, orthophosphate and nitrate followed similar trends during the sampling period from June 1974-January 1975, indicating the sample locations were in a common ground-water flow system.

Two pens were higher in electrical conductivity, calcium, chloride, bicarbonate, nitrate, ammonium and chemical oxygen demand than a third pen or the pasture (Table 3). The pen concentration difference could be attributable to age since two of the pens have been used for 15-20 years, while the third pen is relatively new. Also the two pens with highest concentrations have had the largest stocking rate. Sulfate, potassium and carbonate concentrations were low in both the pens and pasture with no significant difference between pens and pasture. Concentrations of ammonium and chemical oxygen demand were found to be significantly high in all of the pen samples for December 1974 (Table 4). This could be attributed to the high rainfall commonly occurring during November which would be sufficient to saturate the manure pack and result in increased infiltration into the ground-water system. Subsequent natural sealing of the manure pack may have caused the decrease for the month of January. The soil beneath the feedlot is sandy which could allow the passage of the ammonium ion. The high chemical oxygen demand may have been caused by soluble organic matter being carried by the ground water. Nitrate concentrations were fairly consistent (below 10 ppm) and suggest sources other than the feedlots. Sodium and salinity hazards of the ground water beneath the feedlots would be considered low with respect to agricultural standards for irrigation waters.

Bacteria counts were high initially but reduced to nearly zero towards the end of the testing (Table 5). This indicated a possible contamination problem in the installation of the piezometers during the initial stage of the project.

Ground water beneath Feedlot A was only 2-3 feet below surface in a gravelly-sand river bottom material. On the basis of these short-term data it appeared the feedlot had an effect on the quality of the ground water in the immediate vicinity of the feedlot. Since transmissivity and

Table 3. Mean Chemical Concentration of Ground Water Beneath Feedlot A (June 1974—January 1975)

Chemical Parameter	*Pen 5 (4 samples)*	*Pen 2 (5 samples)*	*Pen 6 (5 samples)*	*Pasture (10 samples)*
pH	7.6	7.5	7.5	7.8
Electrical Conductivity (m-mho's/cm)	0.74	1.11	0.52	0.67
Sodium Absorption Ratio	2.2	4.1	1.6	2.9
Calcium (ppm)	44.0	48.0	33.0	27.0
Sodium (ppm)	62	112	41	72
Chloride (ppm)	80	91	34	56
Bicarbonate (ppm)	246	381	224	243
Orthophosphate (ppm)	0.12	0.17	0.10	0.23
Nitrite (ppm)	0.002	0.037	0.002	0.028
Nitrate (ppm)	3.2	3.9	2.3	2.8
Ammonium (ppm)	3.9	4.5	1.5	0.3
Chemical Oxygen Demand (mg/l)	80.9	4.1	2.8	1.6

Table 4. Ammonium Concentrations and Chemical Oxygen Demands of Ground Water Beneath Feedlot A (December 1974 and January 1975)

	Pen 1		*Pen 2*		*Pen 3*		*Pasture*	
	Dec.	*Jan.*	*Dec.*	*Jan.*	*Dec.*	*Jan.*	*Dec.*	*Jan.*
Ammonium (ppm)	14.8	0.1	13.3	3.8	6.5	0.2	0.5	0.2
Chemical Oxygen Demand (COD) (ppm)	321.0	0.1	8.5	2.4	9.5	0.7	1.2	0.6

thickness of the aquifer were not determined accurately, total contribution of pollutants could not be evaluated.

Within Feedlot B soil profiles were taken to establish vertical movement of chemical constituents. Values given were only those parameters which were solubilized in a soil-moisture extract. The investigators felt that these water soluble parameters were the most significant in consideration of ground-water contamination.

Analyses of nitrate, nitrite, ammonium, and chloride indicated maximum concentrations in the upper portion of the soil profile (Figures 2-5). Nitrates were found in concentrations of up to 66 ppm in the upper 10 feet of the profile but decreased to 20 ppm in the 6-7-foot level indicating denitrification was occurring in the profile and significant nitrogen had not penetrated deeply into the soil profile (Figure 2). Nitrites reduced from concentrations of 3-8 ppm in the upper foot to less than 1 ppm below the 3-foot depth (Figure 3). Ammonia concentrations were from 45-88 ppm in the upper 5 feet of profile but reduced to less than 3 ppm at the 5-foot depth (Figure 4). Chloride concentration decreased from highs of over 500 ppm in the upper 1 foot of soil profile to less than 200 ppm at the 8-foot depth and below (Figure 5). One pen had chloride concentration significantly higher (100-200 ppm chloride) than the other pen sampled (less than 50 ppm chloride). This was possibly the result of several factors: (1) difference in age of pens, one being 30 years older than the other; (2) different stocking rate; (3) different management practices between the pens; and (4) different soil and hydraulic conditions.

Similar decreases with depth were noticed for soluble cations including sodium, potassium, magnesium and calcium (Figure 6). The highest was the pH value of 8.3 in the upper 2 feet of the profile but was found to decrease to a pH value of 7.0 at the 4-foot depth (Figure 7).

Table 5. Coliform Concentrations in Ground-Water Samples Beneath Feedlot A (June 1974—January 1975)

Sample	*June*	*July*	*Aug.*	*Sept.*	*Oct.*	*Nov.*	*Dec.*	*Jan.*
Pen 5	0	0	—	—	—	—	0	0
Pen 2	3950	2200	—	—	10	—	65	0
Pen 6	100	17	—	—	0	—	0	0
Pasture 1	0	0	—	—	0	—	0	0
Pasture 2	500	56	—	—	0	—	0	0

Since ground-water levels within this feedlot were over 80 feet from surface and ground-water samples were not analyzed, the results of the soil chemistry indicated the probability of ground-water degradation was low. None of the various chemical constituents analyzed were present in high concentrations in the soil below the 23-foot depth.

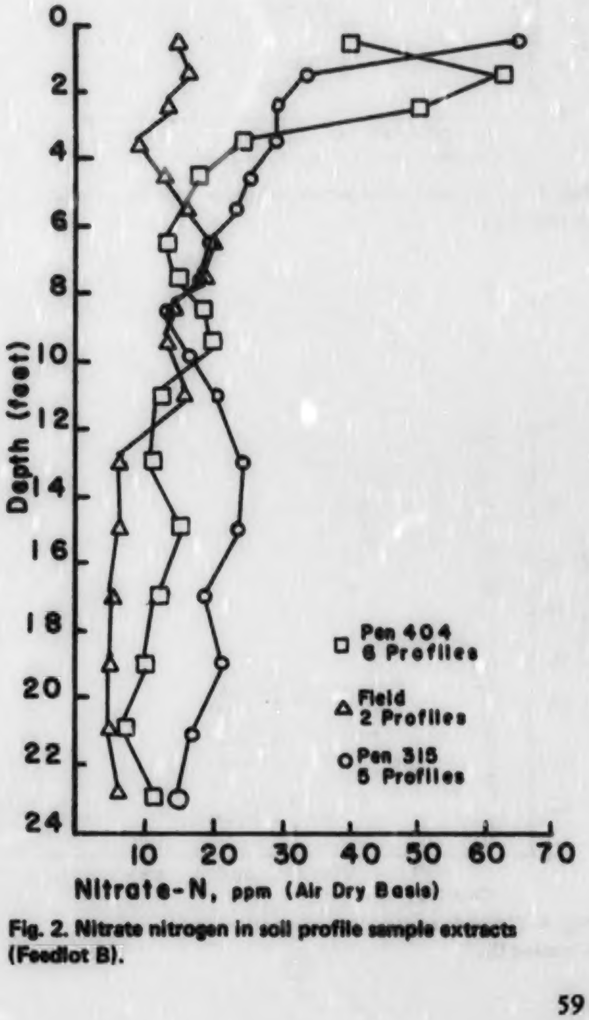


Fig. 2. Nitrate nitrogen in soil profile sample extracts (Feedlot B).

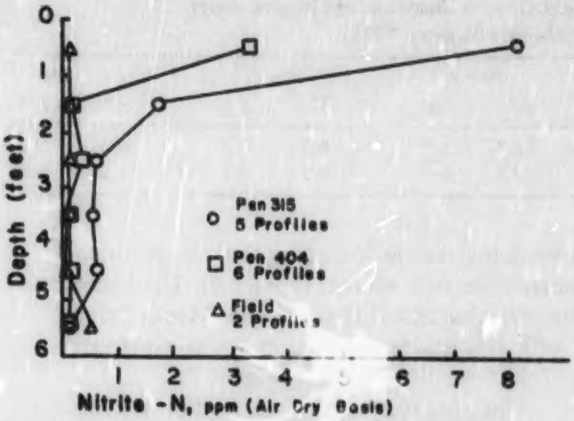


Fig. 3. Nitrite nitrogen in soil profile sample extracts (Feedlot B).

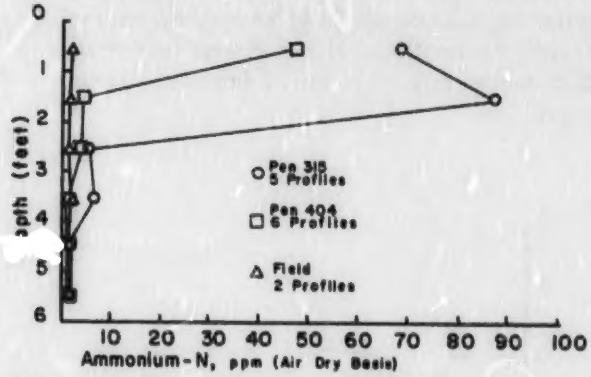


Fig. 4. Ammonium nitrogen in soil profile sample extracts (Feedlot B).

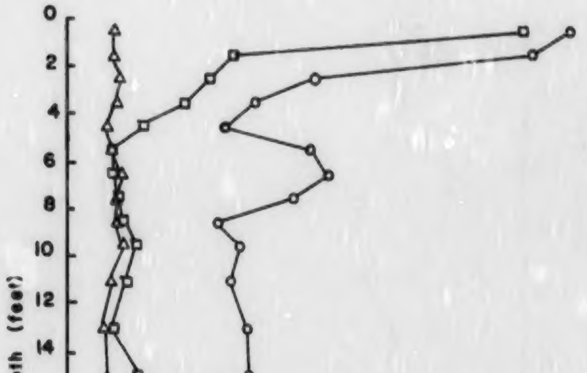


Fig. 5. Chloride content of soil profile sample extracts (Feedlot B).

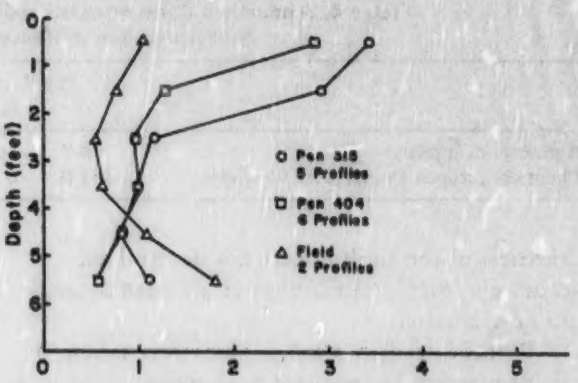


Fig. 6. Total soluble cation content of soil profile sample extracts (Feedlot B).

Land Disposal of Feedlot Wastes

A portion of the study was to determine the effect of land disposal on soil conditions and ground water under different loading rates. An area was chosen near Feedlot A which was typical of the entire floodplain area adjacent to the Boise River. The soil was a coarse sandy loam with water table at a depth of 3-4 feet below surface.

Feedlot manure from Feedlot A was applied to each of 16 separate plots according to Figure 8. The field was irrigated with a furrow-type system also typical of irrigation practice in the area. Piezometers were installed on plots 1-4 and 9-12 and at strategic points surrounding the field to monitor background-water quality. With this

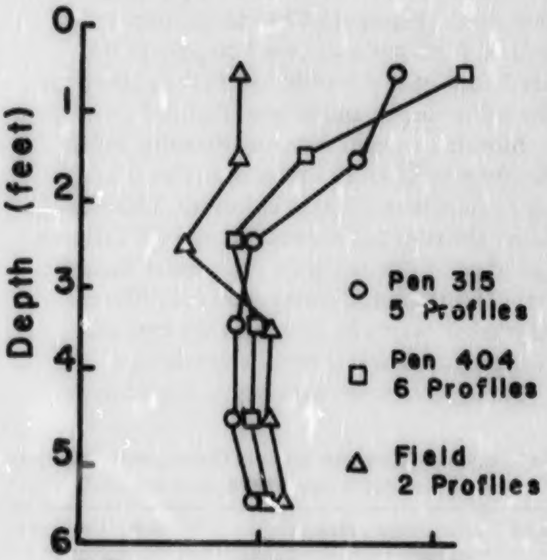


Fig. 7. Observed pH values of profile sample extracts (Feedlot B).

Table 6. Ground-Water Cation-Anion Concentrations Beneath Land Disposal Area*

Plot	Cations (ppm) Ca^{++}	Mg^{++}	Na^{+}	K^{+}	Anions (ppm) HCO_3^-	Cl^-	$SO_4^=$	pH	EC m mhos cm
1	16	10	49	34	260	26	26	8.0	0.55
2	17	9	54	39	260	25	32	8.0	0.53
3	27	14	97	28	337	50	44	7.9	0.80
4	29	15	66	8	277	39	44	8.0	0.67
9	38	20	172	32	417	96	77	7.9	1.30
10	36	19	267	7	423	128	116	8.0	1.59
11	27	14	107	3	304	64	42	8.0	0.84
12	27	15	114	8	307	73	58	8.0	0.89
Background									
Pl-0	26	14	92	5	267	50	57	7.8	0.74
Pl-14	14	7	33	3	173	19	3	8.0	0.37
Drain									
D-2	26	14	108	8	352	37	27	8.0	0.81

* Samples = averages of 18-20 samplings taken from August 1973 through October 1974.

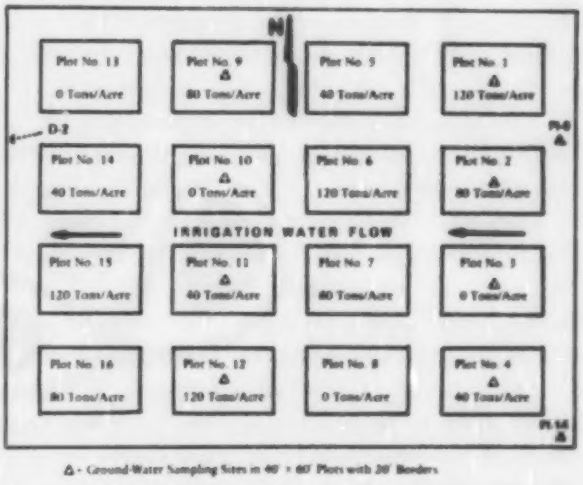
network of sampling points, it was possible to contrast the influence of manure loading at each disposal site with over-all ground-water quality of the site. Samples were taken from August 1973 through October 1974 and data represent mean values based on at least 18 analyses (Table 6).

High sodium values in several of the plots correlated to initial high Na values for these plots prior to application of manure. The high E.C. and HCO_3 values correlated well with the corresponding sodium values. The over-all cation-anion data did not indicate undue contamination of the ground water when compared to the background samples. The Cl and Na data indicated a direct linkage with the ground-water system but compared to the quality of the background samples (PL-0 and PL-14), the ion levels did not indicate massive transfer of ions to the ground-water system.

Nitrogen and phosphorus concentration in the ground water (Table 7) indicated low levels of NO_2^-, NO_3, and P over most of the plots. NH_3 values were constant over the plot and indistinguishable from background samples.

Concentrations of NO_2^- and NO_3^- above background concentrations were detected below plots 9 and 10. The practice of furrow irrigation together with soil type may have had an effect on the distribution of these anions. The high P values in plots 1 and 2 were probably the result of the highly permeable sandy-gravel soil beneath these plots.

Fecal coliform data taken before and after irrigation indicated a direct transfer of fecal coliform into the ground-water system, especially where the highly permeable sandy soil was con-



Δ - Ground-Water Sampling Sites in 40' × 60' Plots with 20' Borders

Fig. 8. Plot layout design for land disposal area (Feedlot A).

Table 7. Observed Nitrogen and Phosphorus Levels in Ground Water Beneath Land Disposal Area*

Plot	NO_2^-	NO_3^-	NH_3	P
1	.01	3.35	1.0	1.64
2	.01	2.35	1.0	1.30
3	.01	3.83	1.0	0.59
4	.01	1.94	1.0	0.05
9	0.15	10.17	1.0	0.20
10	0.11	11.50	1.0	0.24
11	.01	3.20	1.0	0.08
12	.01	2.79	1.0	0.09
Background				
Pl-0	.01	3.36	1.0	0.35

* Mean values obtained on 18-20 samples from August 1973 through October 1974.

Table 8. Influence of Irrigation Water Flow on Fecal Coliform Levels in Ground Water Beneath Land Disposal Area (August 1974)

	No. Fecal Coliforms/100 ml	
Plot	*Before Irrigation*	*After Irrigation*
1	0	2.5×10^4
9	0	1.0×10

(Irrigation make-up water contained from $1.85 - 7.4 \times 10^2$ fecal coliforms per 100 ml.)

cerned (plot 1) (Table 8). A much lower fecal coliform count was found with the heavier silty clay loam test site (plot 9).

Furrow irrigation also had an influence on the transfer of selected anions NO_3, PO_4, and Cl into the ground-water system (Table 9). In the sandy soil (plot 1) all the anions showed an increase in ground-water concentration following irrigation. Plots with heavier soils (plot 11) exhibited little anion movement to the ground-water system.

Mapping

A program was initiated to map the Boise Valley and recommend areas where feedlots and associated land disposal operations could be safely located based upon the previously discussed data on feedlot and land disposal loading rates.

Feedlots

Several factors must be considered in location of feedlots. Some of the physical parameters are slope of the land, flood-prone areas, and areas of near-surface bedrock. These parameters can be defined fairly accurately and mapped accordingly.

The figures in the following text present various generalized zones to be used as planning tools in selection of feedlot sites. They are not hard-fast boundaries but depict areas where problems of various magnitudes may occur. Figure 9 of this section shows the following: (1) slopes over 12%, (2) floodplain area, and (3) bedrock within 5 feet of surface.

Table 9. Influence of Irrigation Water Flow on Ion Movement to Ground Water Beneath Land Disposal Area (September 1974)

	Before Irrigation			*(ppm) After Irrigation*		
Plot	NO_3^-	$PO_4^{\equiv}$	Cl^-	NO_3^-	$PO_4^{\equiv}$	Cl^-
1	2.10	1.06	7	5.20	8.28	41
9	7.20	0.20	84	6.00	0.02	84
3	3.70	0.22	71	3.10	0.28	56
11	3.00	0.01	59	3.00	0.01	56

Excessive Slope

Excessive slopes may cause problems to feedlot operations in several ways. In areas of steep slopes (over 12%), the impermeable organic mat described earlier will not develop. The organic material will be washed or carried to the bottom of the slope rather than developing on the slope itself. This will allow infiltration of nutrients and bacteria possibly resulting in contamination of a ground-water supply.

Excessive slopes also involve the operation of the catchment basins located downslope from a feedlot which are required for containment of surface runoff. The steeper slopes generally make the design and construction of catchment basins more expensive and management of these basins more difficult.

Slopes in excess of 12% were considered to be poorly suited for reasons previously mentioned. Figure 9 shows generalized areas in the Boise River Valley where excessive slopes occur. Excessive slopes occur north of the Boise River in both Ada and Canyon Counties. Scattered areas of excessive slopes also may be found in southern Canyon and Ada Counties along terraces and valleys near the Snake River.

Floodplain

Location of feedlot sites within a floodplain is hazardous because of possible inundation by flood waters and subsequent transport of pollutants into surface waters. Floodplain locations should not be considered for any long-range operation. The expense and problems, both physical and environmental, caused by flooding can be monumental.

The floodplain plates were prepared utilizing data from the United States Geological Survey. Figure 9 shows the approximate area which could be flooded in a probable one-hundred-year event. This area covers most of the lower valley area of the Boise River and certain tributaries of the Boise River. Channel alteration and artificial structures such as bridges could change the flood hazard area.

An important factor relating directly to surface water is the location of a feedlot near a stream. A feedlot should be designed so the catchment basins and settling ponds can be constructed between the surface stream and the

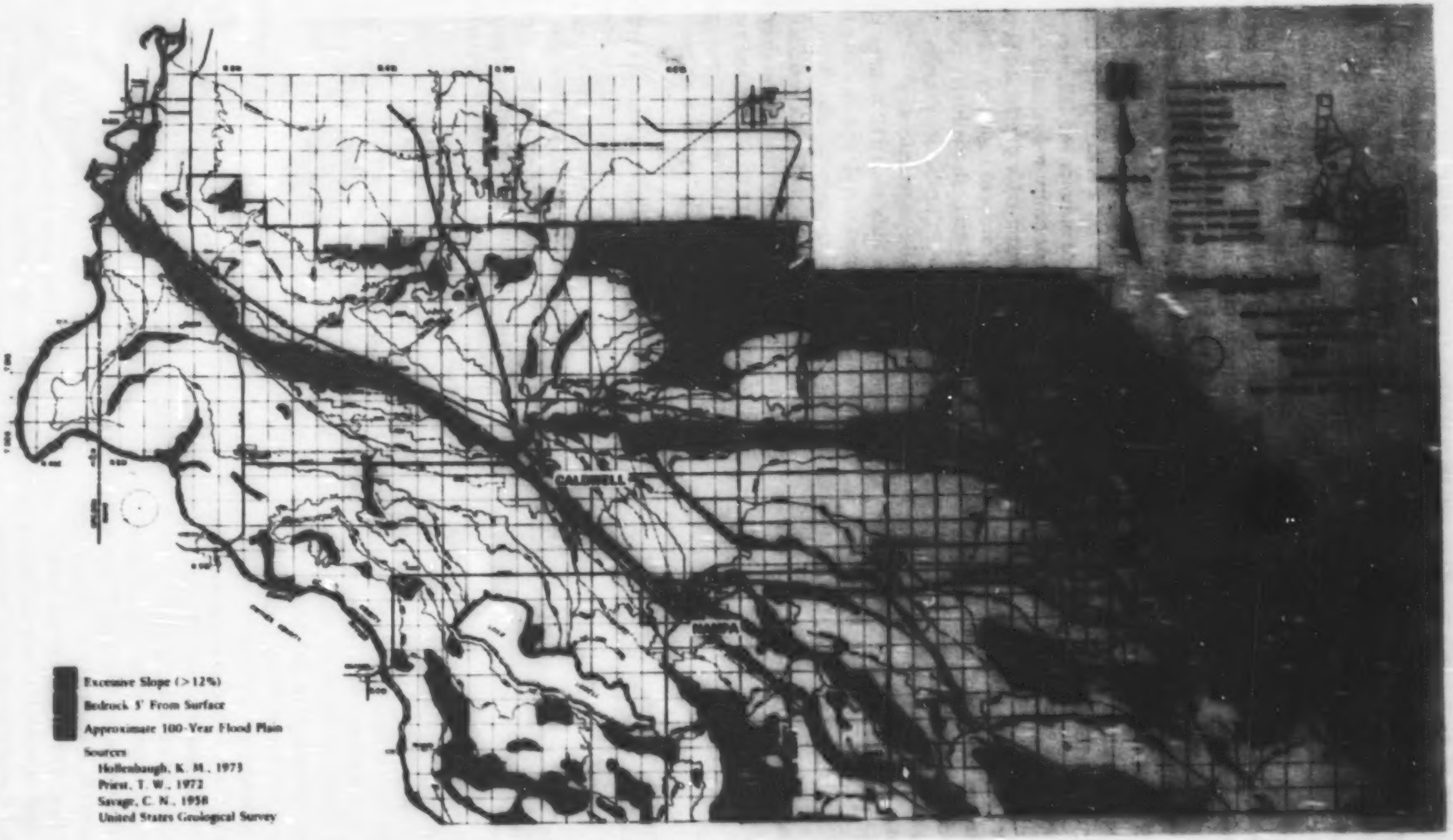


Fig. 9. Cattle feedlot and land disposal hazard areas—excessive slope, near-surface bedrock and flood-prone areas.

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feedlot itself to prevent direct runoff and ground-water seepage from entering the stream. This distance is variable depending on the nature of the stream, terrain, soil types, and feedlot design.

Bedrock Areas

Figure 9 shows near-surface bedrock areas. Outcrops of bedrock, mainly basalt (lava rock), can be hazardous because of the incomplete formation of the impermeable organic mat over the bedrock. If this mat is not formed, surface waters carrying organic and bacterial pollutants may flow into the fractures of the bedrock and could ultimately contaminate a ground-water supply.

The Figure depicts bedrock within 5 feet of the surface. Most of the area of concern which has shallow bedrock is located in southern Ada and Canyon Counties. Because of the scale, the Figure should be used as a guide with detailed geologic investigation performed on specific sites within the shaded area.

Land Disposal of Feedlot Waste

Land disposal of feedlot waste should be given serious attention when locating the feedlot site. First the site should be selected where the economics of disposal provide maximum benefit and where the problems of disposal are best resolved. Analysis of land disposal operations indicates that four factors are of primary importance: (1) depth to ground water, (2) soil permeability, (3) near-surface bedrock or caliche, and (4) proximity to high population area.

Depth to Ground Water and Soil Permeability

Depth to ground water has been found to be one of the most serious limiting factors in disposal operations when viewed in conjunction with soil permeability (Figures 10 and 11). Depths to ground water reach a minimum in the Boise River Valley in mid-July and persist through mid-October. The rise in ground water in the Boise Valley is due to recharge of the ground-water system by irrigation. In planning a land disposal operation, ground-water depths during the late summer and fall months of high water table should be considered.

Shallow ground-water depths over much of the Boise River floodplain are less than 5 feet, and in numerous places the water table is within a few inches of the surface. The shallow ground-water system is also found in several terrace and bench areas. One extensive area extends in an east-west direction from the western edge of Boise to Caldwell. Other shallow zones are located between Caldwell and Nampa and around Lake Lowell (Figure 10).

The shallow ground-water areas should be avoided for land disposal if soils in the area have a medium to high permeability. Highly permeable and medium permeable soils have been mapped and are shown in Figure 10. These soils have a permeability of greater than 6.3 in/hr for the highly permeable soils and between 2.0 and 6.3 in/hr for the medium permeable soils. Where water-table depths are less than 5 feet from the surface during late summer and fall, areas of high and medium permeability soils should be avoided. Where water-table depths are between 5 and 10 feet, the areas of high permeability soils should be avoided. In both the highly permeable soils and medium permeable soils careful water management should be practiced.

Low permeability soils (0.2 in/hr) are limiting because of the amount and type of waste which can be applied. These soils will absorb only a minimum amount of liquid waste material before surface runoff occurs. For this reason they should be reviewed with respect to the size and type of land disposal operation being considered and irrigation water carefully regulated to prevent surface runoff from occurring.

The soil permeability maps are generalized and should be used only for planning purposes, and it should be understood that soil types are complex and often discontinuous. For this reason, any given land disposal site should have a qualified soil scientist or geologist check the soil permeability and ground-water levels prior to the initiation of the disposal operation.

Hardpan

Hardpan zones near the surface may limit the amount and kind of waste material which may be applied in a land disposal operation. The hardpan zones should be considered if the waste material is in a liquid form or if irrigation water is applied to the disposal site. Since hardpan zones are essentially impermeable, the water cannot percolate downward through the zone, and the soil above the hardpan layer may become saturated resulting in surface runoff. This could result in a serious limitation on the amount of waste or the kind of waste which is being considered for a land disposal operation. Hardpan areas are found throughout the Boise Valley; therefore, subsurface investigations for hardpan should precede any land disposal operation.

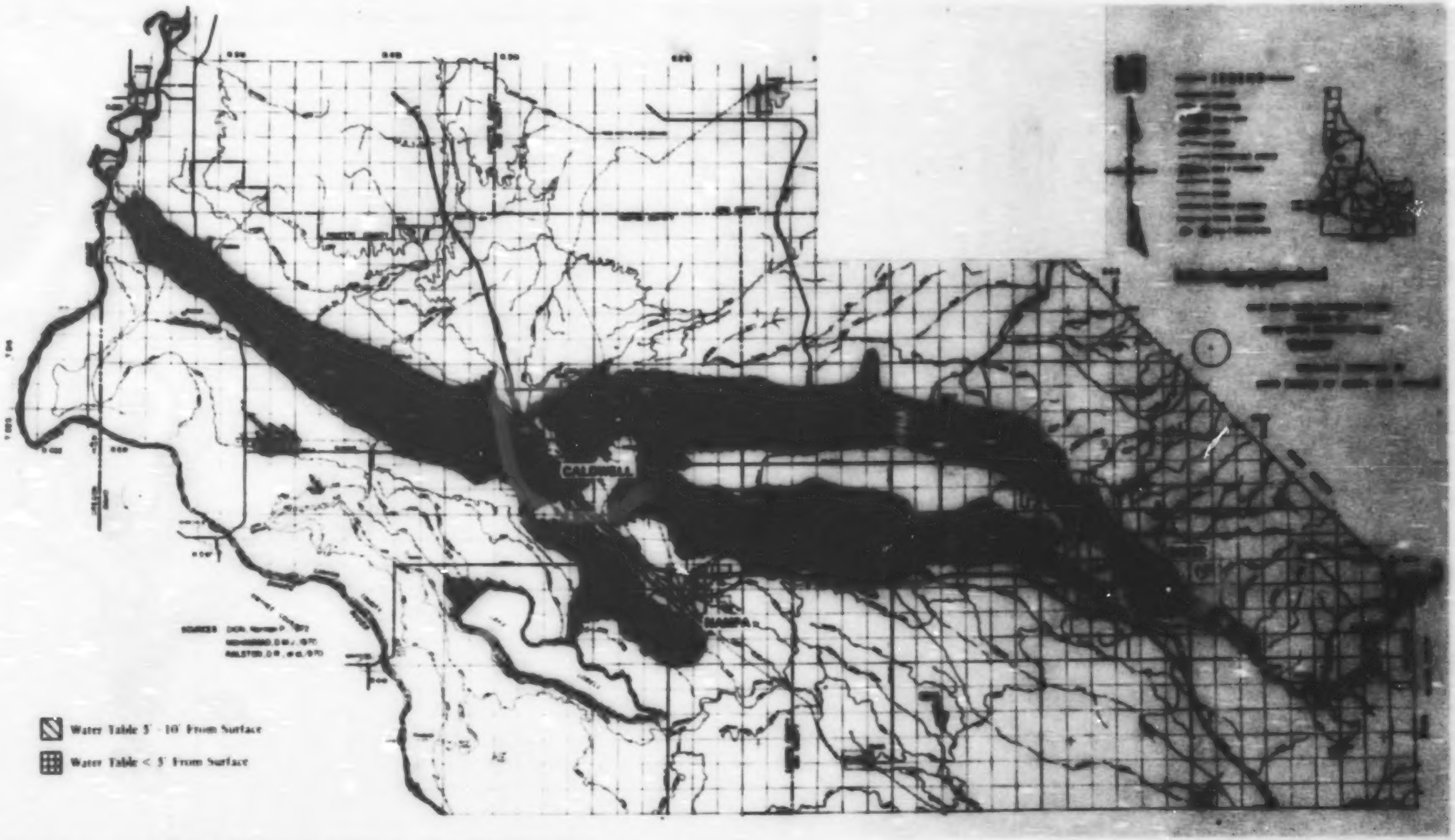
CALDWELL
NAMPA
▨ Water Table 5' - 10' From Surface
▦ Water Table < 5' From Surface

Fig. 10. Cattle feedlot and land disposal hazard areas—depth to water.

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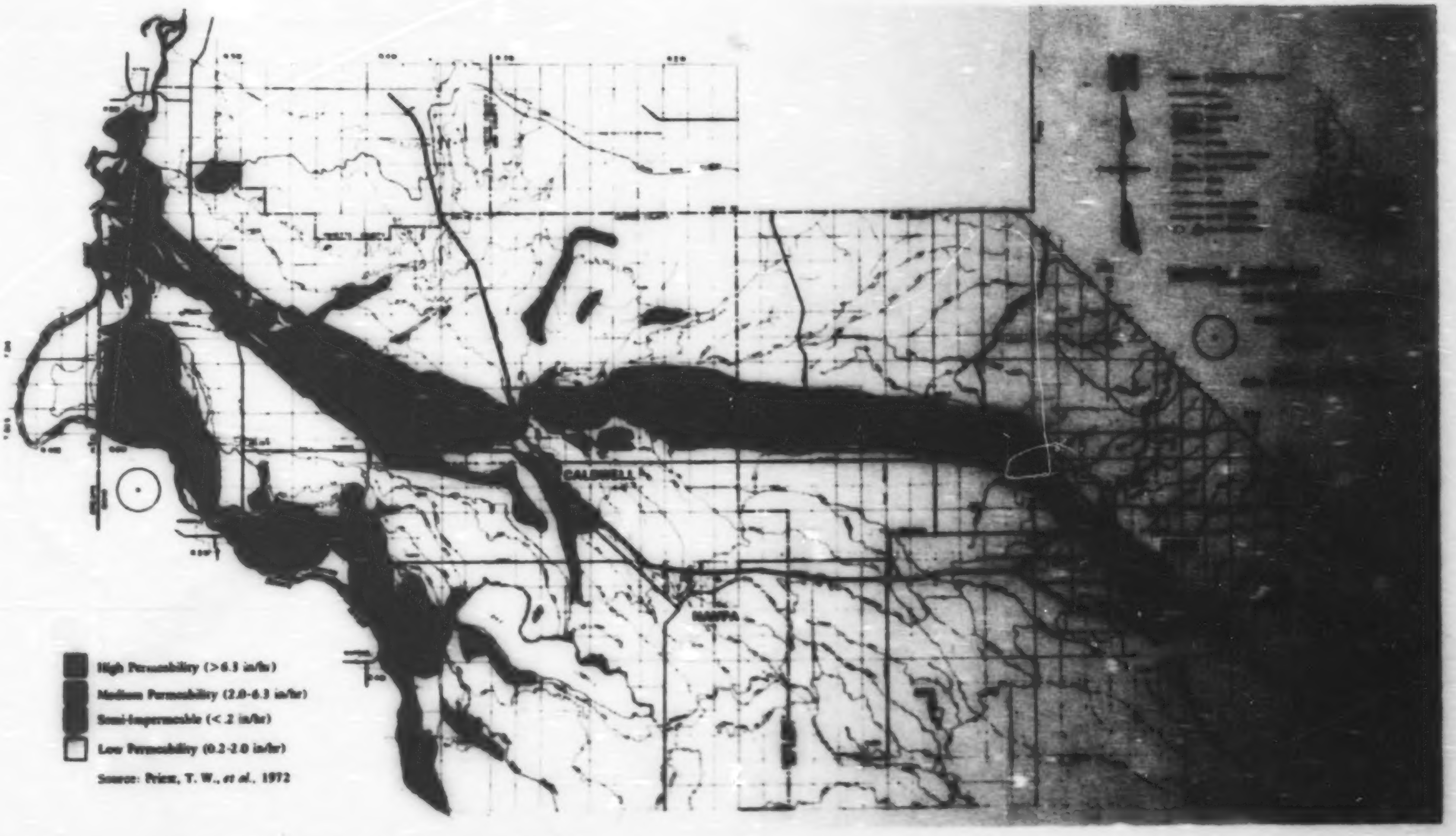
CALDWELL
NAMPA
High Permeability (>6.3 in/hr)
Medium Permeability (2.0-6.3 in/hr)
Semi-impermeable (<.2 in/hr)
Low Permeability (0.2-2.0 in/hr)

Source: Priest, T. W., *et al.*, 1972

Fig. 11. Cattle feedlot and land disposal hazard areas—soil permeability.

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All areas of surface bedrock should be avoided for land disposal sites. Waste material can be transmitted through fractures in the rock to the ground water without any purification by the soil, thus resulting in a seriously contaminated ground-water aquifer (Figure 9).

Urbanization

Population pressure is probably one of the most unpredictable and potentially serious problems in feedlot location and related land disposal operations within the Boise River Valley. Odor and general nuisance complaints stand as the problems. With increased urbanization, through growing cities and scattered high-density subdivisions, odor and general nuisance complaints become a serious matter in feedlot and land disposal location.

Odor from livestock production can be at a maximum during times of manure disposal on land. In order to minimize odor production during disposal operations, it is important that attention be given to weather conditions which may affect the transportation of odors. Of particular concern is the temperature inversion condition which prevents normal air movement and ventilation. The National Weather Service in many areas prepares an Air Pollution Dispersal Forecast which indicates the degree of inversion existing or expected. Feedlot operators should curtail manure disposal operations when the dispersal forecast is rated as poor.

SUMMARY AND CONCLUSIONS

Many factors influence the siting of a feedlot and land disposal operation. With adequate knowledge of the factors and limiting conditions, many of the hazards associated with feedlot operations can be eliminated or greatly reduced. It is also important to realize each sector of the country possesses different characteristics which make application of common environmental standards difficult. Different characteristics can often be found even within State and local boundaries causing difficulties in the use of common guidelines. The previous discussion and following recommendations relate the semiarid Boise Valley region and are based on an interaction of several parameters.

Within the feedlot the manure pack and especially the semipermeable organic mat should be maintained to reduce infiltration and minimize ground-water degradation. Slopes of from 2-3% are recommended to minimize ponding within the feedlot, but slopes over 12% should be avoided because of erosion and incomplete formation of the organic mat. Basalt outcrops should be avoided because of potential ground-water contamination through open fractures. Feedlot location on floodplain areas may be hazardous to animals and property although in certain circumstances, feedlot design can alleviate problems due to flood waters entering the feedlot area.

Land disposal of feedlot waste should be avoided in areas where there exists soil of high permeabilities in conjunction with high water tables (less than 5 feet). In areas of deeper water tables the manure should be incorporated into the soil as soon after application as possible. Sprinkler irrigation is recommended over furrow irrigation in all land disposal operations. Yearly waste loading rates of 40-50 tons/acre are recommended and should be adjusted in accordance with soil permeability, depth to ground water, and irrigation practices. Applications should be made during late fall and the surface allowed to dry prior to incorporation into the soil to decrease potential ground-water pollution due to fecal coliform. If spring application is needed, application should be carried out 3-4 weeks prior to the first irrigation.

Surface runoff from a feedlot area should be controlled through the use of narrow, steep, fast-flowing ditches and channeled to lined settling ponds prior to entering the storage area. The contaminant ponds should be emptied as soon as possible to prevent pond discharge and to reduce odor problems. Disposal of settled solids from the pond should be accomplished in accordance with land disposal guidelines. For both feedlots and land application operations, sites should be selected outside areas of projected future urbanization with careful consideration given to effects of odor and esthetics.

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DISCUSSION

The following questions were answered by L. L. Mink after delivering his talk entitled "The Selection and Management of Feedlot Sites and Land Disposal of Animal Waste in Boise Valley, Idaho."

Q. *What is the nitrate level of the ground water under those cultivated areas next to the feedlots? Are they higher than those waters beneath the feedlots?*

A. Mean nitrate-nitrogen concentrations of ground water beneath a pasture and a cultivated field near feedlot A was 2.8 mg/l and 3.36 mg/l respectively. Mean concentrations of nitrate-nitrogen of ground water beneath three feedlot pens were 3.2 mg/l, 3.9 mg/l, and 2.3 mg/l. This indicates nitrate-nitrogen in the ground water beneath the feedlots was not significantly different from the nitrate-nitrogen beneath the cultivated areas.

Q. by H. Bouwer. *Your results seem to contradict others where old feedlots that were not cleaned contributed less nitrate to ground water than feedlots regularly cleaned. This was due to ammonium accumulation which increased pH. This in turn decreased nitrification and increased ammonia volatilization. Comment?*

A. All feedlots within this investigation were being utilized prior to and during the investigation. The slightly greater amounts of nitrates found in the older pens is thought to be due to accumulation of nitrates over time. If a pen is cleaned to a degree the organic mat is disturbed or removed, increased infiltration would result and a definite increase in nitrates could be expected. Although the data indicate higher concentrations of ammonia beneath the older pens there was no significant difference in pH in the ground water beneath the pens. This could be due to the low concentrations of both nitrates and ammonia, therefore not creating the conditions of increased ammonia volatilization and decreased nitrification.

Q. *With respect to nitrates in soil systems beneath feedlots, studies in other areas have shown high nitrate concentrations subsequent to feedlot abandonment and deterioration of the manure pack. What have been your observations in Montana subsequent to feedlot abandonment?*
A. We did not specifically look at abandoned feedlots during this investigation. Observations since the investigation indicate that upon abandonment the impermeable organic mat deteriorates allowing surface water to infiltrate through the organic material. The percolating waters carry substantial amounts of nitrates which cause high concentrations in ground waters beneath the feedlot.

Q. by W. Turner. *How long does it take to develop an effective manure pack?*
A. No testing was performed during the study but observations on newly established feedlots indicate a period of from 3 to 6 weeks is required to establish an effective organic mat.

Q. by Don Runnells. *Were the analyses for the components below the "dry" feedlot performed on the soils or on* vadose interstitial waters? *In either case, how did you sample the materials?*
A. The chemical analysis in the vadose zone was performed on air-dried soil samples and the values reported are those parameters which were solubilized in a soil moisture extract. The samples were taken utilizing a small bucket-type hand auger with the samples collected in plastic bags and taken directly to the laboratory for analyses.

Q. by P. K. Saint. *Your feasibility map would recommend feedlot location on areas with thick vadose zones. But these are zones of regional recharge and would you not contaminate large areas of ground water? Low water-table areas are ground-water discharge areas, and would these not be more suitable?*
A. The recommended areas for feedlots did not consider depth to water. As long as an effective organic mat is maintained, data indicate ground-water pollution from feedlots will be minimal. Soil profile data indicates the cation-anion concentrations decrease with depth so at a ten-foot depth the concentrations are at or near background levels. Since we did find above background amounts of several constituents beneath feedlots where the water table was from 2 to 5 feet below surface, we recommend feedlots be located in areas where water-table depths are at least 5 feet from surface and preferably over 10 feet from surface. To locate feedlots in ground-water discharge areas would cause problems with surface-water systems since the ground water would be discharging in surface-water systems within a short distance.

Q. *Have you considered the rising trend of the water table in the area in delineating the feedlot locations and selecting the recommended depth to the water table?*
A. Yes, the data used to delineate areas of high water tables were gathered during late summer and early fall. This is during the period when the water table is at its maximum elevation due to irrigation in the Boise Valley.

Q. by K. Childs. *How important is the ability of differing soil types for waste assimilation? Was this considered? Don't low permeabilities increase retention time for wastes and consequently increase treatment of waste? Are waste constituents removed with the zone of saturation?*
A. As evidenced by the data from the land disposal portion of the study, soil types play an important role in waste assimilation. High concentrations of bacteria, nitrates, nitrites, and chlorides were detected in the ground waters beneath plots which had high permeable sandy soils but no significant increase of these parameters was detected in lower permeable silt loam soils. Within the study we did not attempt to determine the waste assimilative capability of the zone of saturation.

Q. by Maxine Goad. *What consideration did you give to the availability of suitable sites for impoundments to catch surface runoff? Did you investigate the possibility of ground-water contamination caused by seepage from such impoundments?*
A. We recommended areas of steep slopes be avoided because of problems associated with design and maintenance of catchment facilities. We also recommended an adequate area be maintained between a surface stream and the feedlot for surface runoff catchment facilities. The study did not address the problem of seepage from surface-water impoundment facilities. Within the feedlots investigated such facilities were lined with an impermeable liner.

Q. by Clinton C. Whitmer. *What percentage of the Boise Valley was left available to feedlot use?*
A. A large area within the Valley appears to be suitable for feedlots as far as physical restrictions. Only flood-prone areas near the Boise River and areas of excessive slopes along the Boise Ridge are not suitable. The major problem is that of urbanization—considering the scattered development which is now occurring in the Valley one may conclude there is no acceptable safe location.

Q. by Don Runnells. *Is there any legal or regulatory machinery in Idaho to implement (or impose) the recommendations of work such as yours? Are there any significant efforts to preserve your agricultural lands?*
Q. Within Idaho the protection of agriculture lands is in the hands of the county zoning agencies. Within the Ada-Canyon Counties comprising the Boise River Valley a serious attempt is being undertaken to protect agricultural lands from urbanization but at the present time is plagued by local politics and special interest groups. The planning and zoning agencies are also working to implement the recommendations of the study. We are very pleased the local cattle feeders association has expressed considerable interest in the results and recommendations of the study.

"The Lycoming County, Pennsylvania, Sanitary Landfill: State-of-the-Art in Ground-Water Protection," by M. Todd Giddings, Jr. appears on page 125.

Prediction of Future Nitrate Concentrations in Ground Water[a]

by C. P. Young, D. B. Oakes and W. B. Wilkinson[b]

ABSTRACT

Over the last few years rises in the nitrate content of ground water from wells and springs in the principal aquifers of the United Kingdom have been observed. In a number of cases the concentrations have exceeded the WHO lower recommended limit. In order to determine the reason for the rise, to assess whether it will continue and the eventual nitrate levels, the Water Research Centre has undertaken an extensive programme of drilling and sampling on the Chalk and Bunter Sandstone, and by August 1976, twenty-two sites had been examined. This work has established that high nitrate concentrations (peaks up to 60 mg/l NO_3-N have been observed) are present in the unsaturated aquifers at fertilized arable/ley sites. At unfertilized grassland sites nitrate concentrations are low (less than 4 mg/l NO_3-N) and below fertilized established grassland values are in the intermediate range. At one farm site near Winchester, models to predict the rate of movement of nitrate through the unsaturated and saturated Chalk have been developed. These suggest that the nitrate levels at this site will remain at an essentially constant value of about 4 mg/l NO_3-N until the late 1970's when they will rise progressively to about 14 mg/l NO_3-N. The models have been checked against tritium data and the approach is now being extended to other sites.

1. INTRODUCTION

A rise in the concentration of nitrate in ground water has been observed in many countries over the last few years and is a cause of concern to health authorities and water engineers. It has been recognised for about 30 years (Comley, 1945) that excessive quantities of nitrate in drinking water can cause methaemoglobinaemia in infants under the age of 6 months; the baby develops a grey-glue colour which has been called "water well cyanosis." The condition is extremely rare but if untreated, permanent damage or death may result. There is some evidence to suggest that increased methaemoglobin levels in the blood of older children may result from the ingestion of "high nitrate" water (Petukhov and Ivanov, 1970) but adverse effects on health have not been proved. This paper describes some of the techniques being used to estimate the future changes in the concentration of nitrate in ground water in the United Kingdom.

Ground water is a major source of public supply in England and Wales where it meets about 30% of the demand. In some areas, particularly southeast England, it is the major source. The quality of ground water pumped from the two principal aquifers in the United Kingdom, the Chalk and Bunter Sandstone (Figure 1), is

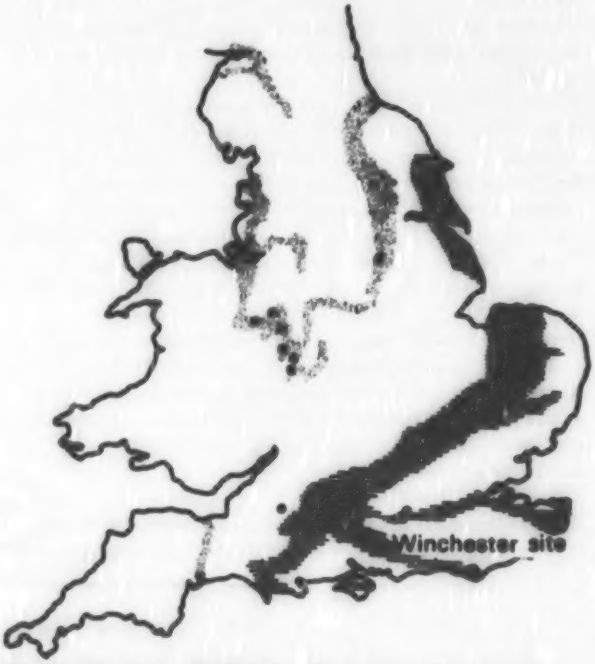

Fig. 1. Sites of the Water Research Centre nitrate investigations.

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

[b] Water Research Centre, Medmenham Laboratory, P.O. Box 16, Henley Road, Medmenham, Marlow, Buckinghamshire SK7 2HD, England.

generally exceptionally good requiring only minimal treatment. However, increasing nitrate levels in both aquifers have been observed (Davey, 1970; Greene and Walker, 1970; Satchell and Edworthy, 1972; Foster and Crease, 1974; Young and Hall, 1976). At the present time ground water from more than 100 public supply sources either continuously or intermittently exceeds the World Health Organisation lower limit for nitrate of 50 mg/l.

The WHO recommend in their European Standards for drinking water that nitrate should be less than 50 mg/l but suggest that an upper limit of 100 mg/l is acceptable provided that health authorities are notified if the lower limit is exceeded so that problems arising with infants may be readily recognized. (Nitrate analyses are usually expressed in terms of nitrogen and from here on, this is adopted in this paper. The equivalent of the WHO limits of 50 and 100 mg/l are 11.3 and 22.6 mg/l nitrogen.)

The water industry is clearly concerned to know whether

(a) the source of nitrate can be controlled,

(b) the trend of increasing nitrate concentrations will continue and the level to which they will eventually rise,

(c) currently acceptable ground-water sources are also going to be affected,

so that alternative water sources may be planned, treatment methods developed, or other appropriate action taken as necessary.

Research was required in order to answer these questions and in October 1974 the Water Research Centre expanded its programme in this area.

2. SELECTION OF SITES AND INVESTIGATION TECHNIQUES

The principal objectives of the field studies have been to

(1) measure the amount of nitrate contamination in the zone of aeration and in ground water at typical sites on the Chalk and Bunter Sandstone;

(2) determine the relationships between the above measurements and meteorological, hydrogeological, historic land use, and agricultural factors;

(3) assess whether nitrate in moving down through the zone of aeration and in the aquifer, is subject to denitrification; and

(4) estimate the future concentrations of nitrate in ground water, initially on a detailed local level, but proceeding to a catchment scale.

The objectives are being met by an extensive programme of drilling and sampling on the Chalk and Bunter Sandstone, and to date 22 sites have been investigated (Figure 1).

Sites have been selected to cover a range of land-use situations. In order to ensure compatability they have been located on outcrop areas free from superficial deposits. An unsaturated zone depth of at least 20 metres has been required so that the rate of downward movement of contaminants through this zone could be determined. Detailed records of land use and fertilizer application rates over a period of at least 10 years, but preferably 30 years, together with meteorological records have been considered essential. The Ministry of Agriculture, Fisheries and Food (MAFF) and the water authorities have given invaluable assistance with site selection and the supply of the basic data.

Great care has been taken in the choice of drilling techniques so as to obtain fully representative samples of the rock and pore fluids. Procedures employing a fluid to flush rock particles from the drilling bit could not therefore be used. Dry augering and drive coring were the methods generally adopted to obtain samples. However, under certain adverse conditions in the Bunter Sandstone air flush rotary drilling was employed (Gray *et al.*, 1976).

Continuous augering at 114 mm (4.5 in) diameter at depths not exceeding 30 m (98 ft), with sample collection from the auger flights, was used at the preliminary stages of the investigations to give a broad nitrate profile through the unsaturated zone and to determine the suitability of the site for more detailed study. If the site was suitable, a drive coring or a rotary drilling method was used to give 100 mm (4 in) diameter continuous undisturbed core. These boreholes ranged in depth up to 200 m (656 ft) and in some cases passed through the unsaturated zone and deep into the saturated aquifer. All samples, on removal from the borehole, were immediately sealed in double polythene wrappings and stored in insulated containers for transport to the Centre's laboratory. Pore water was extracted from the rock samples using a high-speed centrifuge (Edmunds and Bath, 1975) and was analysed for nitrate and nitrite by Auto Analyser techniques. Samples for tritium analysis were passed to another organisation's laboratory who undertook the preparation of the samples and the measurement on a contract basis.

3. GENERAL RESULTS OF FIELD INVESTIGATIONS

The literature describing the leaching of nitrogen from soil under a variety of land use and meteorological conditions is profuse. However, measurements in the United Kingdom and elsewhere have generally been restricted to the soil zone (Wild and Babiker, 1975). Nitrate profiles to a depth of up to 30 m (98 ft) have been measured in granular aquifers in Israel and the United States (Olsen *et al.*, 1970; Adriano *et al.*, 1971; Reinhorn and Avnimelech, 1974; Smith and Young, 1975; Mielke and Ellis, 1976; Lund *et al.*, 1976), but as far as the authors are aware the results of the present study describe for the first time detailed nitrate profiles in the unsaturated zones of consolidated aquifers.

The 22 sites cover a range of land-use situations that may be broadly grouped into three types:

(a) fertilized arable including temporary leys,

(b) fertilized long-term grassland, and

(c) unfertilized long-term grassland.

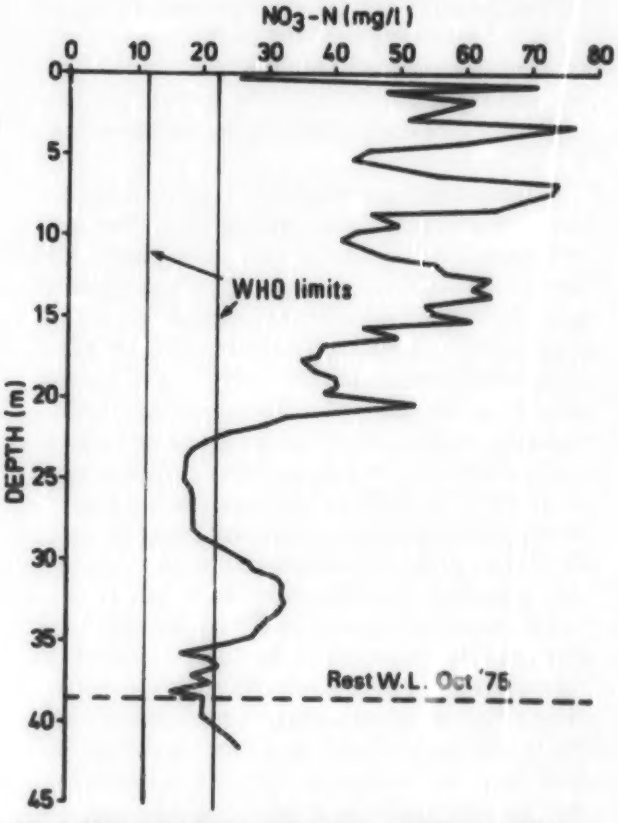


Fig. 2. Measured nitrate profile below a fertilized arable site on Chalk.

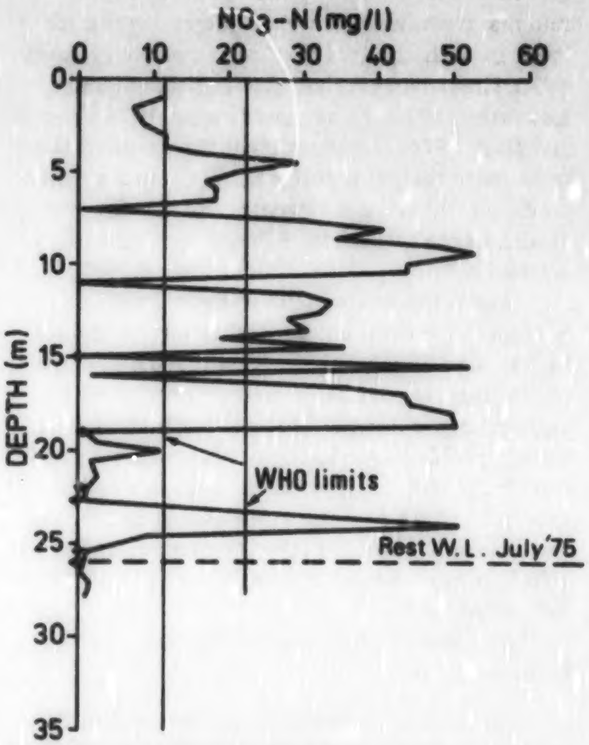


Fig. 3. Measured nitrate profile below a fertilized arable site on Bunter Sandstone.

Nitrate profiles through the zone of aeration beneath inorganically and organically fertilized arable land on both the Chalk and Bunter Sandstone are given in Figures 2 and 3. The profiles show peak nitrate concentrations which are in some cases up to five times greater than the WHO lower recommended limit. The peaks are often irregularly interspersed with zones of low concentration. In contrast, profiles below permanent unfertilized grass show comparatively low levels which are almost always below 4 mg/l NO_3-N (Figure 4). Fertilized long-term grassland sites give profiles (Figure 5) that are intermediate between the arable and unfertilized cases.

Deep drilling and sampling below the water table has also been carried out at a number of sites. At one site on the Bunter Sandstone adjacent to a water-supply well, which has recently recorded an increase in nitrate, a profile with high nitrate peaks has been measured. These results are as yet incomplete and so a full interpretation cannot be made at this stage.

It is clear from the results described above that a considerable quantity of nitrate is present in the zone of aeration below fertilized sites on the Chalk and Bunter Sandstone. However, in order to

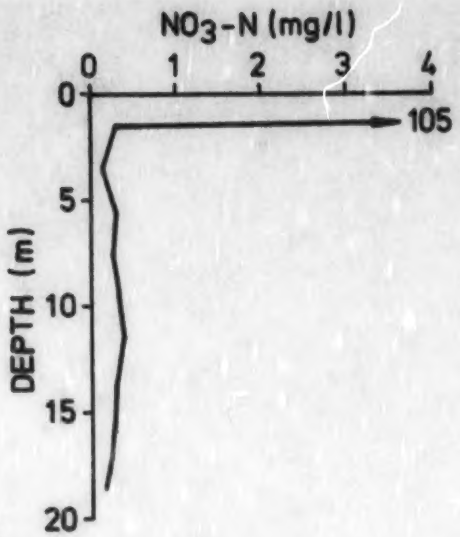


Fig. 4. Measured nitrate profile below an unfertilized permanent grassland site on Chalk.

attempt a prediction of the future concentration of nitrate in ground water it is necessary to establish quantitative information on

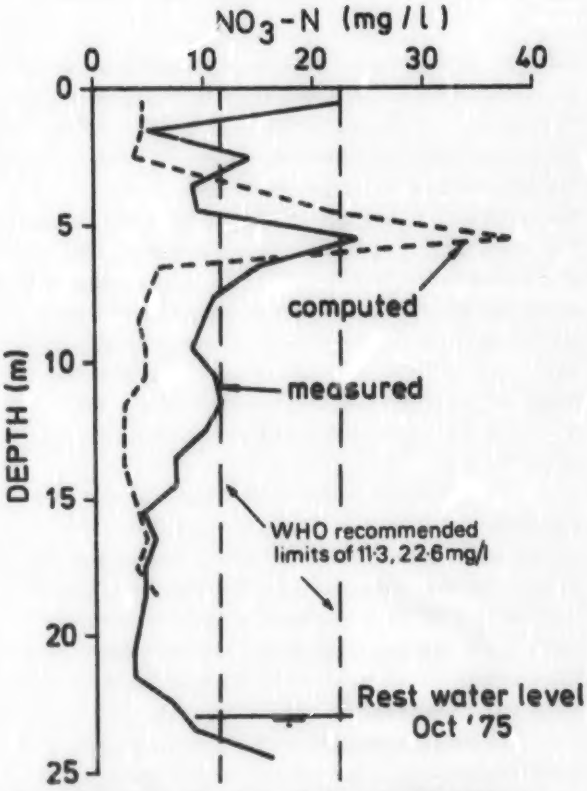


Fig. 5. Measured and computed nitrate profile below a fertilized grassland site on the Chalk—Bridget's Farm Borehole G, Maryland field.

(a) relationships between land use and nitrate leaching loss from the soil,

(b) the rate of downward movement of nitrate, and

(c) the amount of denitrification that is taking place in the zone of aeration.

The results from all of the sites are being analysed in order to establish these relationships. The investigations and analyses at a site on the Chalk near Winchester (Figure 1) are described in detail in the following section.

4. CHALK SITE NEAR WINCHESTER

4.1. Land Use, Hydrogeology and Borehole Locations

Investigations were undertaken on a Chalk site at Bridget's Experimental Husbandry Farm between October 1974 and January 1976 (Young *et al.*, 1976). Detailed land utilisation and agricultural practice records are available from 1948 when the MAFF took over the farm.

The drilling and sampling procedures used in the investigation were described in section 2.

The farm is situated on the south facing slopes of the Itchen Valley (Figure 6). The lowest fields border the River Itchen at a level of about 45 m (148 ft) Ordnance Datum from which the land rises steadily over a distance of 1.5 km (1 ml) to a level of 116 m (381 ft) O.D. at the northern extent of the farm boundary.

The soil cover within the farm boundaries and the surrounding area is generally thin and with a low clay content. It rests directly on the Chalk so that, except under conditions of very heavy rainfall, surface runoff is negligible. Estimates of infiltration from 1948 to the present have therefore been made on the basis of rainfall less actual evaporation.

Because the farm is experimental in nature, changes in land use do not necessarily follow the national trend which has shown a rapid increase in arable acreage over the last 40 years. The local increase for the Parish of Itchen Valley is shown in Figure 7. There has also been a tenfold increase in fertilizers over the country as a whole during this period.

The ground-water catchment in which the farm is situated is comprised of Upper Chalk and it is estimated that its base lies at about Ordnance Datum beneath the farm site. The beds at the site are almost horizontal with a regional dip of about 1° to the south.

The Chalk is a fine-grained carbonate rock traversed by horizontal and near vertical joints and fissures. The jointing pattern is very irregular with block sizes ranging from a few centimetres to more than a metre. The size of the openings is difficult to estimate. Horizontal discontinuities up to 10 mm (0.4 in) in width have been reported by Ward *et al.*, 1968, but Foster and Milton, 1974, have noted tightly closed joints at relatively shallow depths. Near the water table these discontinuities have been enlarged by solution and this leads to the high transmissivity values that are associated with the Chalk aquifer. The rock matrix is composed of fossil debris ranging in size from 1 to 100 μm and pore diameters are generally less than 5 μm (Figure 8). The intergranular permeability is therefore low, of the order of 1×10^{-3} m/d (2×10^{-2} gals/d/ft^2), but the porosity is high at 0.3 to 0.4.

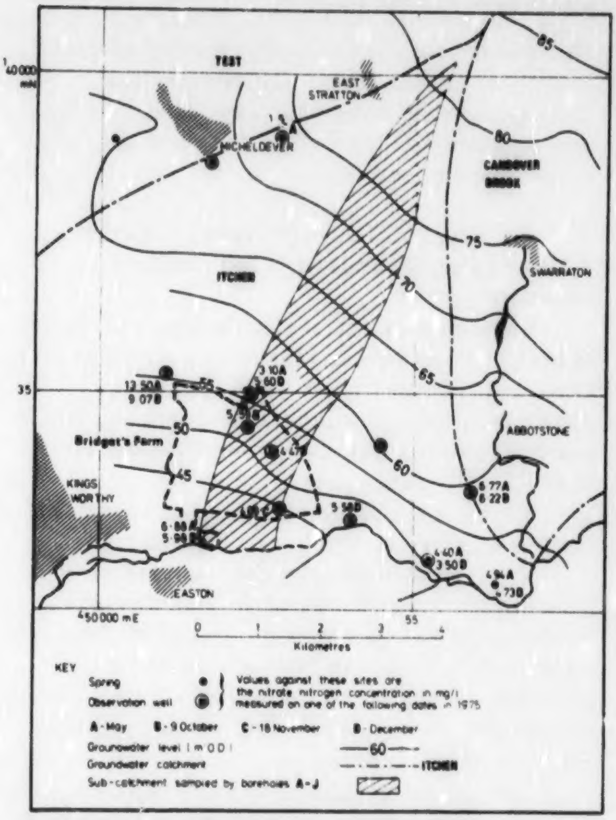


Fig. 6. Location of Bridget's Farm, rest water levels, ground-water catchment and nitrate concentrations in ground water and springs in 1975.

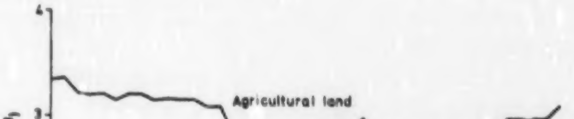


Fig. 7. Changes in land use—Itchen Valley parish.

Fig. 8. Scanning electron micrograph of the Chalk matrix.

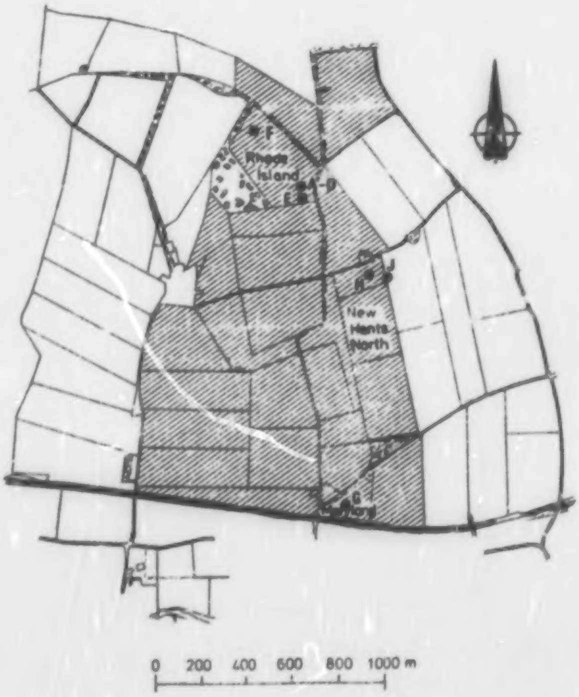
Ground-water levels in the Chalk of the area are shown in Figure 6. These are based on the information collected during the investigation and data previously assembled by the Southern Water Authority (1975). Long-term ground-water level records are not available at the farm site but fluctuations of up to 3 m (10 ft) have been recorded at the head of the catchment.

Nitrate determinations were made on water samples collected from boreholes and springs in the area and these results are plotted in Figure 6. The values generally lie between 3 and 7 mg/l NO_3-N. No trend with respect to location is apparent. The nearest public water-supply well is on the south side of the Itchen Valley and this has shown a rise in the nitrate concentrations from about 3 mg/l NO_3-N in 1955 to 4.5 mg/l NO_3-N in 1975.

During the course of the investigations 8 boreholes, designated A to H in Figure 9, were sunk by the Centre through the unsaturated and in some cases into the saturated Chalk. At the same time a 76-m (249-ft) deep observation borehole was sunk at location J by MAFF in connection with the experimental storage of animal waste slurry in unlined Chalk pits (Chumbley *et al.*, 1976).

Boreholes A to D were drilled during the period October 1974 to January 1975 and boreholes E to F in October 1975. All are situated in Rhode Island field (Figure 9) which since 1948 has been rotated between cereals and kale with periods of ley. Details of land usage together with nitrogen fertilizer applications are given in Figure 10. Borehole F was located in a part of the field which had only been cleared of woodland in 1951.

Borehole G (Figure 9) was drilled in Maryland field during November 1975. This field was in continuous use as rough grazing from 1949 to 1969 followed by ploughing and the growing of spring barley in 1970 and 1971, since when it has been returned to ley (Figure 11). Borehole H was drilled in New Hampshire north field during October 1975, land-use records being shown in Figure 12.



KEY Boreholes ●A

▨ Fields for which detailed land use and fertilizer records have been tabulated are shaded

Fig. 9. Location of investigation sites and boreholes on the Bridget's Farm site.

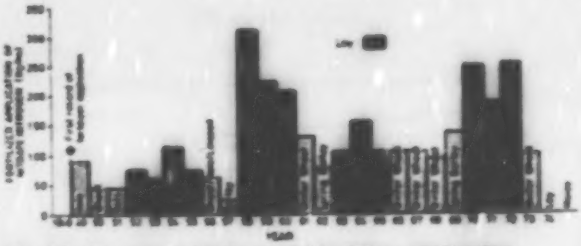


Fig. 10. Changes in land use and fertilizer application 1946-1974, Rhode Island field—Boreholes A to F.

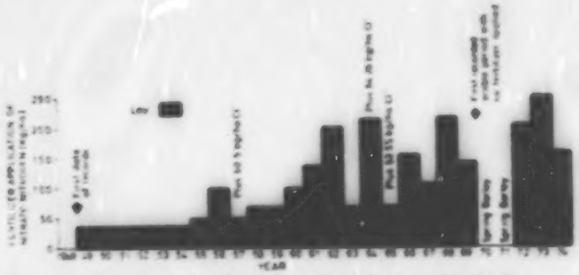


Fig. 11. Changes in land use and fertilizer application 1948-1974, Maryland field—Borehole G.

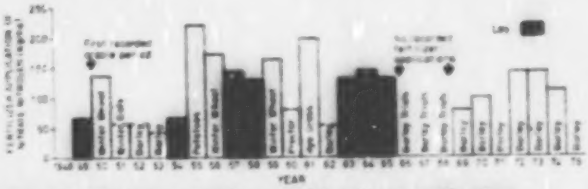


Fig. 12. Changes in land use and fertilizer application 1948-1974, New Hampshire north field—Borehole H.

4.2. Nitrate, Chloride and Tritium Profiles

The change in concentration of nitrate with depth below ground surface is plotted for boreholes D, E, F, G and H in Figures 13, 14, 15, 5 and 16 respectively. All show peak values that exceed the WHO 11.3 mg/l NO_3-N recommended limit. Boreholes D and E were sited within 10 m of each other in the same field and while the general shape of their profiles is similar, the integrated nitrate concentration to a depth of 29 m (95 ft) in borehole E is 32% greater than that in borehole D.

Chloride determinations were only made on the pore water from borehole G (Figure 17). The source of the chloride is attributable to the appli-

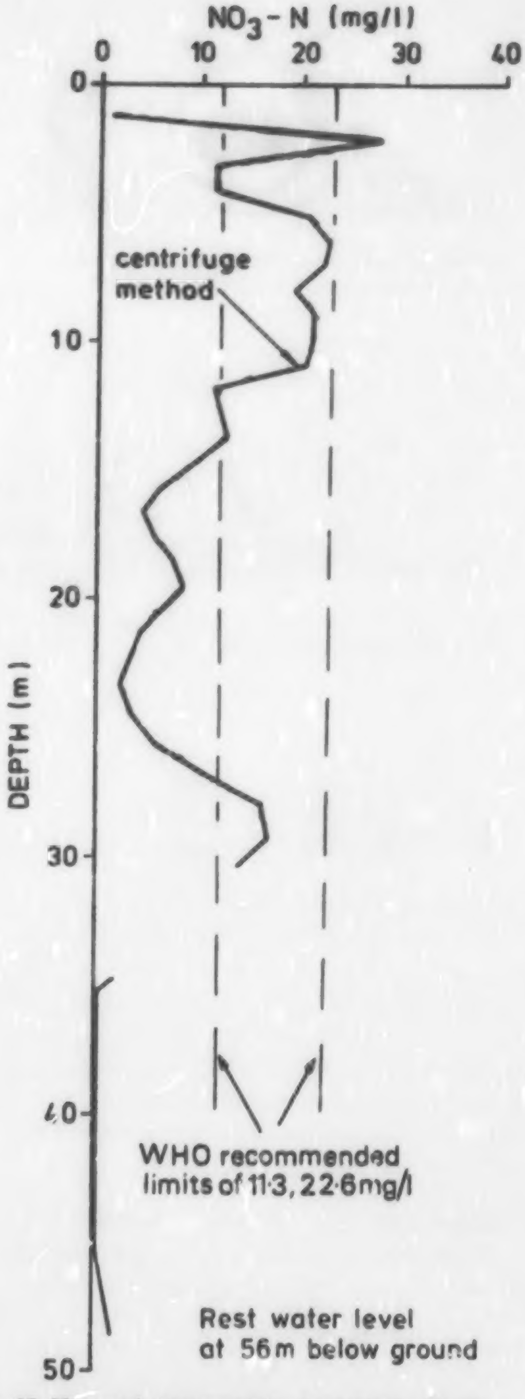


Fig. 13. Measured nitrate profile—Borehole D—Rhode Island field.

cation of potassium chloride fertilizer in 1957, 1964 and 1965.

The concentration of tritium was measured on chalk samples from boreholes A, B and C. All show very similar profiles and as borehole C is the most complete, it is presented in Figure 18.

4.3. Unsaturated Zone Flow Model

The characteristics of the Chalk have been described in section 4.1. Moisture content determinations on samples of unsaturated Chalk

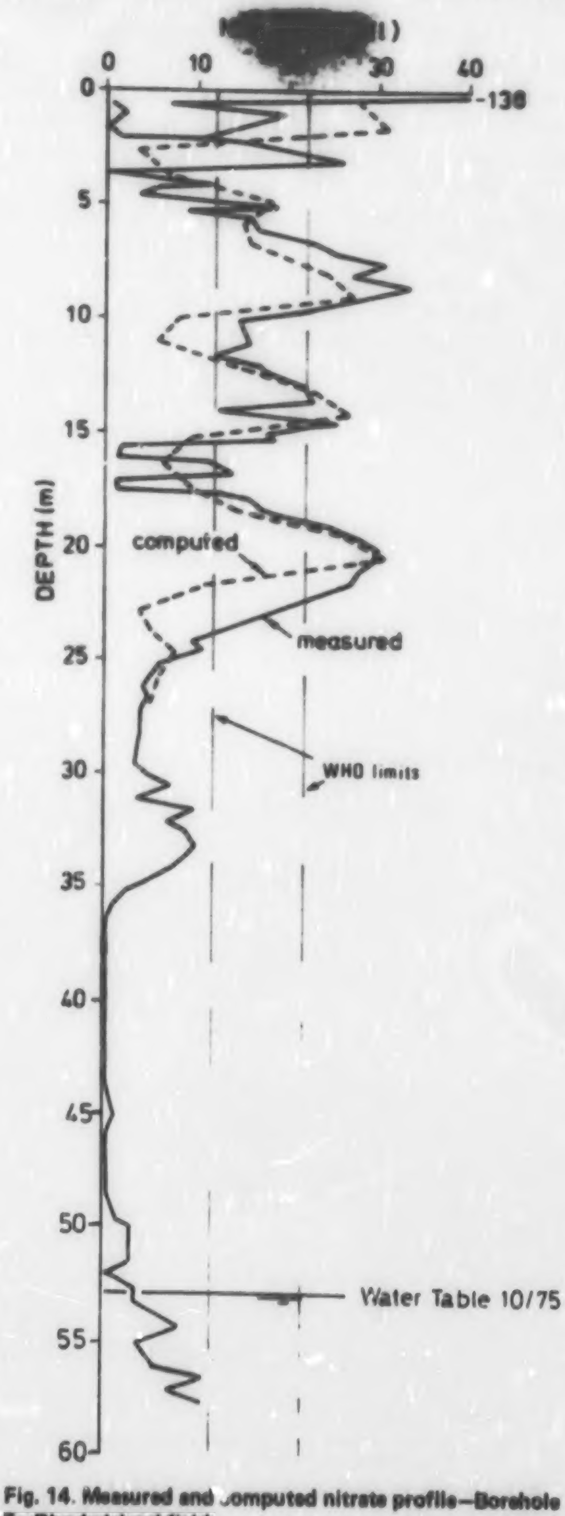


Fig. 14. Measured and computed nitrate profile—Borehole E—Rhode Island field.

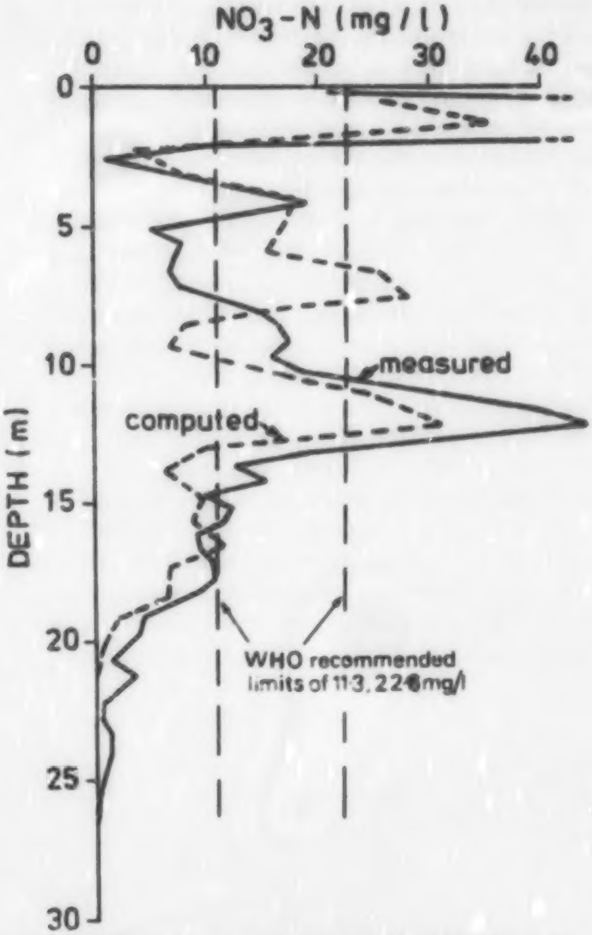


Fig. 15. Measured and computed nitrate profile—Borehole F—Rhode Island field.

suggest that the intact blocks between the fissures are almost fully saturated, the water being held by surface tension forces. Water infiltrating from the surface and moving downwards by intergranular flow would migrate at a rate of 1 m/yr if typical values of permeability and porosity are assumed (1×10^{-3} m/d and 0.35). This is a similar rate to that reported by Smith *et al.*, 1970, for the migration of tritium through unsaturated Chalk. However, it is difficult to discount the multitude of fissures that are so evident in Chalk exposures, as having no place in the flow process. These fissures are the main flow channels in the saturated zone, are responsible for the aquifer's high transmissivity, and give a 1 to 2% effective porosity.

It has been suggested (Foster, 1975; Young *et al.*, 1976) that relatively rapid infiltration through the unsaturated zone may take place along joints and fissures and that solutes may diffuse between the moving water in the joints and the relatively static water in the matrix. This may account for the observed relatively slow downward movement of tritium and nitrate. It is uncertain whether there would be sufficient time to allow effective diffusion to take place but assuming this, it can be shown that the unsaturated Chalk column behaves in a similar manner to a chromatogram. When equilibrium is established the solute ions divide between the static pore water and the fissure water in the ratio of their respective volumes. It can also be shown that the ratio of the total porosity to fissure porosity is equal to the ratio of the velocity of the water in the fissures to the downward velocity of the contaminant front. The properties of the unsaturated Chalk are such that the infiltrating water would move downwards at a rate some 20 to 40 times faster than the associated contaminants. Sharp input peaks would persist during downward migration provided that rapid equilibrium is attained but some blurring would occur due to dispersion. If fissure flow is sufficiently slow, then nitrate, chloride and

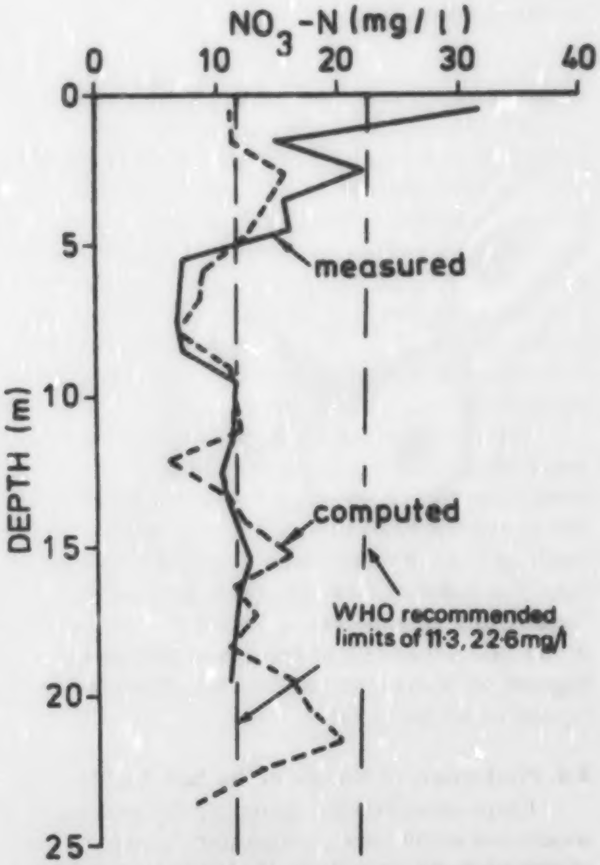


Fig. 16. Measured and computed nitrate profile—Borehole H—New Hampshire north field.

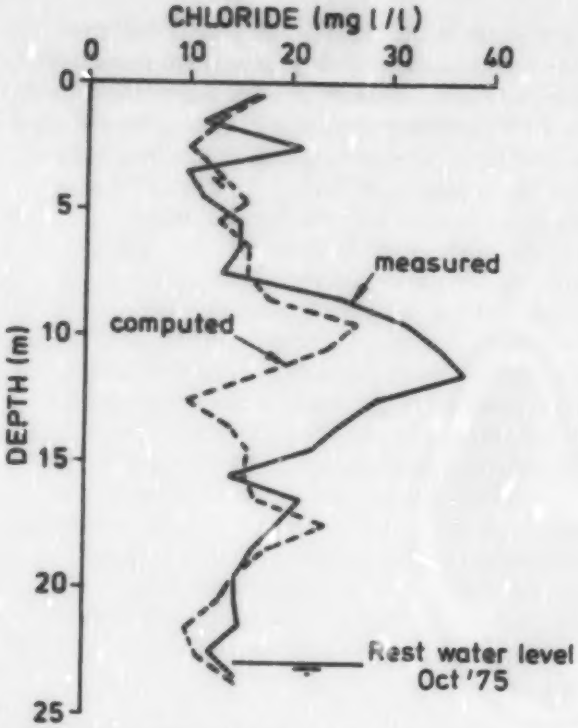


Fig. 17. Measured and computed chloride profile—Borehole G—Maryland field.

tritium fronts would move at very similar rates.

Further experimental work is necessary before the flow mechanism in the unsaturated Chalk is fully understood. However, both the intergranular flow and fissure flow concepts suggest that a uniform velocity, downward-flow model may be used to represent the movement of contaminant inputs from the surface. This simple model was therefore adopted. Some attenuation of the contaminants was allowed for by applying a vertical dispersion factor in the model.

Many wells in the Chalk give a rapid rise in water levels following a period of rainfall. In some cases it has been possible to calculate that this represents 10 to 15% of the infiltration resulting from the rain, and it is believed that the rapid rise is due to flow through large fissures in the unsaturated zone. It has therefore been assumed in the model that 15% of the annual infiltration together with any associated contaminants moves rapidly to the water table.

4.4. Production of Nitrate in the Soil Profile

Crops assimilate nitrate during the growing season and within limits the greater the application of nitrate the greater the yield. A proportion of the vegetable matter is removed on harvesting but the residue of roots, weeds and microflora remains, adding to the soil organic matter. In this organically bound form the residual nitrogen is not available to other plants but must be mineralised by specific soil bacteria which are active from March to December when the soil temperature exceeds 5°C. The mineralised nitrogen is taken up by growing crops during the period March to September but mineralisation continues into early winter during which period precipitation exceeds evaporation so that infiltration through the soil and into the unsaturated aquifer occurs. Nitrate is thus carried downwards with the infiltrating water. Kolenbrander (1975) estimated that for cereals and root crops, 50% of the applied fertilizer becomes available as organic matter for mineralisation. This value was used in the model, and it was further assumed that it could all be leached from the soil. However, not all the material is available in the year of application and on the basis of Kolenbrander's work it was assumed that mineralisation took place over a 3-year period. The

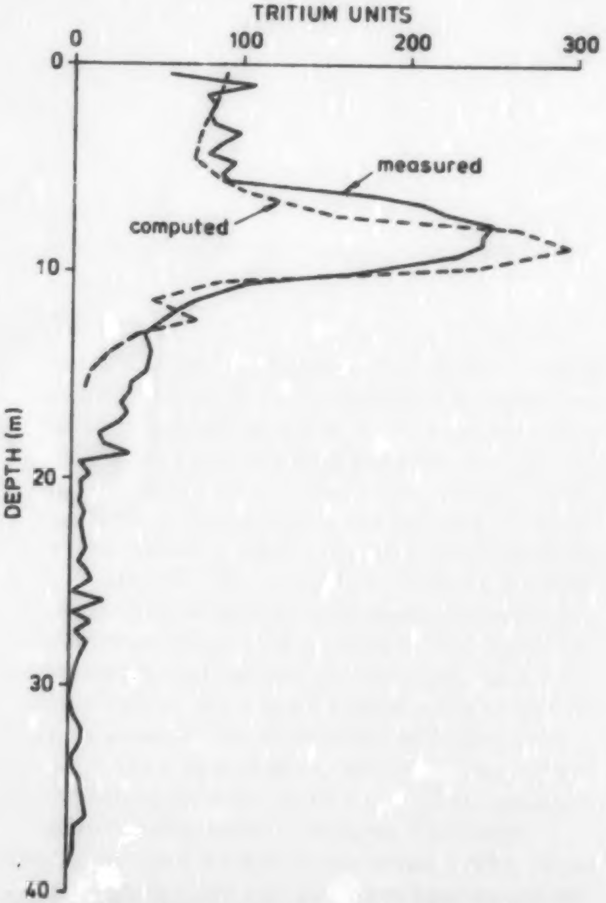


Fig. 18. Measured and computed tritium profile—Borehole C—Rhode Island field.

amount of nitrate available for leaching as a fraction of the application rate is given in Table 1.

Nitrifying bacteria are aerobic and are thus normally restricted to the upper aerated rooting zone of the soil. Compaction under permanent grassland or leys may reduce the thickness of the aerated zone and lead to anaerobic conditions at depth in the soil profile. Anaerobic bacteria in this zone may reduce the nitrate in the infiltrating water to gaseous nitrogen. However, even under permanent grass a small proportion of the mobile nitrate will be leached through the soil zone. Williams (1975) has reported leaching values of about 4 mg/l NO_3-N from below established grass/lucerne leys and this is in accordance with the nitrate profiles obtained for unfertilized permanent grassland sites and shown in Figure 4. This value of 4 mg/l NO_3-N was therefore adopted in the model for the nitrate content in water leaching from established grass.

If a soil is undisturbed, organic material may accumulate and Kolenbrander (1975) has shown that equilibrium between the inputs and losses by mineralisation may only be achieved after a period of between 10 and 40 years. Naturally developed soil profiles are destroyed by ploughing, and if this takes place at the end of the growing season in autumn, the creation of aerated conditions favours mineralisation and leads to nitrate loss. High concentrations of nitrate in drainage water from ploughed grassland have been reported (Cooke and Williams, 1970; Kreitler and Jones, 1975; Reinhorn and Avnimelech, 1974; Olsen *et al.*, 1970).

Table 1. Nitrogen Available from Arable Crops [N (kg/ha) available as a fraction of the fertilizer application rate N (kg/ha)] (1 kg/ha = 0.89 lb/acre)

	Year of application	*1st year following application*	*2nd year following application*
Roots	0.35	0.10	0.05
Cereals	0.25	0.15	0.10

Table 2. Nitrogen Available from Ploughed Grassland

Years in grass prior to ploughing	*N (kg/ha) released by ploughing*			
	Total	*Year of ploughing*	*1st year following ploughing*	*2nd year following ploughing*
1	30	18	9	3
2	60	36	18	6
3	170	100	50	20
4 or more	260	155	80	25

Table 3. Solute Migration Rates

Borehole	*Solute*	*Migration rate (m/yr)*
E	nitrate	1.05
F	nitrate	0.84
G	nitrate	1.00
H	nitrate	1.05
A and B	tritium	1.05
C	tritium	0.80
G	chloride	1.00

The data on the rate of buildup of soil organic matter below grassland is, however, rather sparse but from information based on the work of Smith and Young, 1975, an estimate of the amount of nitrogen released by ploughing grassland of various ages was established and the results are shown in Table 2.

4.5. Application of Model to the Profiles

The land-use records, fertilizer applications, and estimates of infiltration rates for the period 1948 to 1974 were used together with the nitrate release rules in the constant rate flow model to predict the nitrate profiles in the unsaturated zone. The measured and computed profiles for boreholes E, F, G and H are given in Figures 14, 15, 5 and 16.

Using the same model the tritium profiles from 1954 to 1975 were calculated for boreholes A, B and C. A comparison of the measured and computed profile for borehole C is given in Figure 18.

An attempt was made to model the chloride profile obtained from borehole G. The historical information on the application of potassium chloride fertilizer is not very reliable but for the years 1957, 1964 and 1965 is approximately 60, 64, 54 kg/ha Cl. Chloride in rainfall at the Bridget's Farm site was estimated to be 15 mg/l Cl. This data was used in the model and the measured and computed profiles shown in Figure 17.

The downward migration rate for the contaminant used in the model was in all cases adjusted so as to give the best match between the measured and computed data and these values for each profile are given in Table 3. The rates all fall in the range 0.8 to 1.05 m/yr (2.6 to 3.4 ft/yr) and there appear to be no significant differences between solutes.

The authors are unaware of any experimentation on the relative mobility of various ions in the Chalk. However, in other media, similar migration rates for nitrate and chloride have been reported from both field and laboratory studies (Kurtz and

Melsted, 1973; Adriano *et al.*, 1971; Garwood and Tyson, 1973; Wild and Babiker, 1975).

The tritium profiles are particularly important as in this case the surface inputs are accurately known. An excellent comparison between the computed and measured results for both the peak position and size is obtained (Figure 18). The tritium migration rates obtained are in close agreement with those measured by Smith *et al.* (1970) in the unsaturated Chalk of the Lambourn area some 50 km (31 mls) from the Bridget's Farm site.

The computed nitrate profiles, due to the uncertainties concerning inputs, do not match the measured profiles so well as those for tritium. No attempt was made to simulate the measured profile in borehole E below a depth of 28 m (92 ft) as this corresponded to a pre-1948 land-use activity for which data is not available. The results from boreholes F and G suggest that the rules governing the release of nitrate from ploughed land require some refinement.

There are anomalies between the profiles in boreholes D and E. The holes are adjacent to each other and are located in the same field. Nitrate peaks occur at the same depth but the magnitude of the peaks in borehole D are generally less than those in borehole E. The position of the peaks supports the hypothesis of a uniform downward migration rate for nitrate but the differences in peak magnitude suggest that the nitrate inputs from the soil may not be uniform in any one field. This may be due to differences in the soil structure or local irregularities in the Chalk. However, the over-all behaviour of the model in matching the nitrate, chloride and tritium profiles is generally good.

4.6. Future Concentrations of Nitrate in Ground Water — Predictive Models

The concentration of nitrate at any point in the saturated aquifer is a function of the nitrate content of the water draining from the unsaturated zone above that point and of additions from ground water flowing into the area. Vertical mixing in the saturated zone is likely to be slow and inputs of differing concentrations along a flow path may remain distinct for some distance. However, wells and boreholes effectively mix the water over their depth of penetration so that, unless specialised sampling techniques are employed, an average sample is obtained. Pore water samples collected from cores below the water table will reflect the composition at various levels. This may be seen from the profiles of boreholes E and G (Figures 14 and 5) which penetrate 4 and 2.3 m (13 and 8 ft) respectively below the water table. Borehole E in particular shows a series of peaks increasing to 10 mg/l NO_3-N at a depth of 3 m (10 ft) and these are possibly discreet inputs that have occurred at an upgradient situation from borehole E some time in the past. This concept is shown diagrammatically in Figure 19.

If the downward flow model for the unsaturated zone is accepted as being valid, it may be used to predict the rate at which nitrate already in the profile and future surface inputs would enter the ground water. This approach was applied to borehole E by assuming that the past agricultural practice and infiltration at this site will be repeated cyclically in the future. Figure 20 shows the model predictions from 1950 to 2030. From 1950 to 1994 the 4 mg/l NO_3-N entering the ground water is due entirely to rapid fissure flow from the surface. From 1994 onwards the high nitrate concentrations currently present in the profile reach the water table and levels rise to peak concentrations of nearly 30 mg/l NO_3-N in the year 2010. It must be stressed that the results presented in Figure 20 represent the conditions only under a single area of one field. In reality there would be a range of migration rates (Table 3), and this will have the effect of smoothing the nitrate inputs to the water table by lowering the peaks and raising the troughs of the curve shown in Figure 20.

Estimates of the expected nitrate concentrations in ground water were obtained from a catchment model in which infiltrating water and nitrate were routed through the saturated zone of the aquifer. The sub-catchment used in this study is shown in Figure 6, and encompasses most of the area of Bridget's Farm. The sub-catchment was divided into five zones, based on the available records of land use. In three of the zones, land use was identified as being similar to that practised in the Rhode Island field (Figure 10), and in one other zone land use was similar to that practised in the

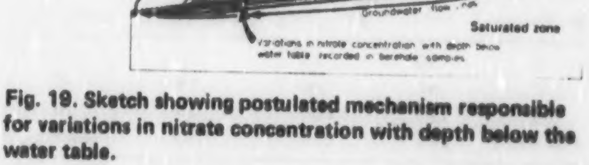

Zones of differing nitrate concentration moving downwards sub-parallel to land surface
Unsaturated zone
Water table
Groundwater flow path
Saturated zone
Variations in nitrate concentration with depth below water table recorded in borehole samples

Fig. 19. Sketch showing postulated mechanism responsible for variations in nitrate concentration with depth below the water table.

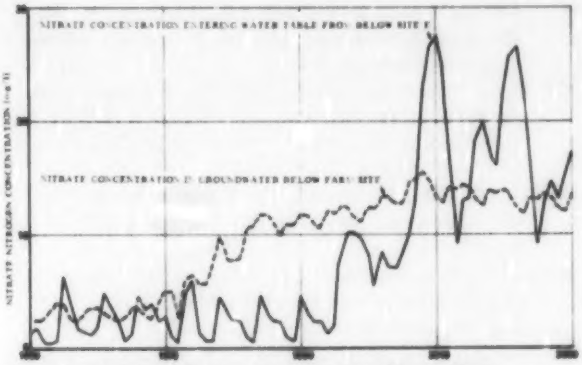


Fig. 20. Model predictions for period 1950 to 2030.

(a) Nitrate concentrations entering water table from below site E.

(b) Average nitrate concentrations in ground water below farm site.

New Hampshire north field (Figure 12). The fifth zone comprised woodland. Mean depth to water table was estimated for each zone. The vertical flow model was used to estimate the nitrate flux across the water table in each zone from 1950 to 2030 with the assumption that past agricultural practice and infiltration rates would be repeated cyclically in the future. It was assumed that there would be a range of downward migration rates through the Chalk in each zone, which results in smoothing of the nitrate inputs to the water table. Analysis of the available data suggested the migration rates were normally distributed with mean 1 m/yr (3.3 ft/yr) and standard deviation 0.07 m/yr (0.2 ft/yr), and the nitrate flux across the water table was calculated accordingly.

A simple catchment model, assuming complete mixing of water and nitrate in each zone, was used to calculate mean concentrations of nitrate in ground water beneath borehole E, and in the spring discharge at the River Itchen. Times of transit through the aquifer were estimated to be small compared with times of transit through the unsaturated zone, having a maximum value of about 5 years. The catchment model predictions of nitrate in ground water beneath borehole E are shown in Figure 20. The estimate for 1975 is 5.7 mg/l which compares well with a measured value of 5.6 mg/l (see Figure 6). The peak value of 15.5 mg/l occurs in 2008 with a long-term average of about 13 mg/l. The steady rise in concentrations from 1970 onwards results from contributions from upgradient zones where the unsaturated thickness is less than at borehole E and the delay times are correspondingly less. The predicted concentrations in the spring discharge rise progressively from 6.7 mg/l in 1975 to about 14 mg/l from 2000 onwards. The measured concentration in 1975 was 6.9 mg/l.

4.7. Denitrification

The predictive models described in section 4.6 assumed that denitrification does not occur once the nitrate has left the soil and entered the unsaturated aquifer. Little direct evidence has been obtained to suggest that significant biological action takes place at depth. However, the carbohydrate content of the Chalk in borehole E has been measured at 0.1 g/kg dry Chalk. This corresponds to a carbon content of 120 mg/l of pore water which, if available as an energy source, could enable denitrification to the extent of about 100 mg/l NO_3-N (Young, *et al.*, 1976) to take place. If reliable predictions are to be made of the future concentrations of nitrate in ground water, it is important to establish whether significant denitrification within the aquifer is taking place, and work on this objective is continuing at the Centre.

5. CONCLUSIONS

1. High concentrations of nitrate, in excess of the WHO upper recommended limit, have for the first time been measured in the interstitial water of the unsaturated and saturated zones at a number of sites on the two principal aquifers in the United Kingdom.

2. Quantitative relationships have been established between the amount of nitrate in the profiles and the historic land use and fertilizer applications. Fertilized arable/ley sites have high concentrations, fertilized permanent grassland sites intermediate concentrations, and unfertilized grassland sites low concentrations of nitrate in their profiles.

3. Detailed analysis at a site on the Chalk near Winchester has led to the formulation of a flow model for the unsaturated zone which, on the basis of the historic land use, predicts nitrate profiles which are in good agreement with those measured. The validity of the model has been checked against measured tritium and chloride profiles.

4. A series of such models has been integrated into a catchment model which predicts that the nitrate concentrations in the Chalk ground water at the Winchester site will rise progressively from its present value of about 4 mg/l NO_3-N, reaching 14 mg/l NO_3-N during the early 2000's. The models have assumed that little or no denitrification takes

place within the aquifer. The modelling studies are being extended to other sites.

6. ACKNOWLEDGEMENTS

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(Mr. Wilkinson's Discussion Questions and Answers appear on page 230.)

The Contribution of Fertilizer to the Ground Water of Long Island[a]

by Joseph H. Baier[b] and Kenneth A. Rykbost[c]

ABSTRACT

In 1973, the Suffolk County Department of Environmental Control in cooperation with Cornell University began a study on nitrogen (N) fertilization of potatoes and turfgrasses. The research and demonstration project has shown that current practices result in substantial N losses; and, in many cases, excessive use of N reduces crop yields and turf quality. Annual N losses of 50 lbs. per acre (55.5 kg-N/ha) are sufficient to cause a concentration in the aquifer's surface layer of 10 mg/l nitrate-N (New York State Drinking Water Standard).

The eastern portion of Long Island supports a productive agricultural industry whose main crop is potatoes. Ground-water surveys have shown that the aquifer system of this area is contaminated with nitrate nitrogen. The average potato grower applies 200 to 250 lb-N/a (222 to 278 kg-N/ha) at planting time; and depending upon a number of factors, N recovered in harvested tubers varies from 75 to 150 lb-N/a (83 to 167 kg-N/ha). Losses to the ground water could vary from 50 to 175 lb-N/a (55.5 to 194 kg-N/ha). The study has shown that the application of 150 lb-N/a (167 kg-N/ha) can still maintain maximum potato yields and keep the N loss to ground water below 50 lb-N/a (55.5 kg-N/ha) by improving nitrogen-use efficiency. This is done through splitting N applications so that one-third to one-half is applied at planting and the remainder is applied prior to the period of rapid crop growth and nutrient uptake. On-farm demonstration plots are being used to convince growers to reduce N rates and adopt more efficient application methods.

The western portions of Long Island are highly urbanized and turfgrasses may be fertilized at rates up to 350 lb-N/a (389 kg-N/ha). Potential leaching losses are high for turfgrass as N is not recovered in harvested plant materials. Experiments are underway to establish rates of biomass N buildup under several fertilization regimes. Preliminary results indicate that N-use efficiency increases with more frequent but smaller N applications. Encouraging the use of low maintenance turf species appears to be the best long-term solution.

INTRODUCTION

Long Island is a rapidly growing area with fixed land and water resources. Water for all uses (industrial, commercial, agricultural and municipal) is obtained from ground water. The quality of this resource is most important and is of particular concern on the eastern portion of Long Island where a limited quantity of water exists. Agricultural areas are pointed to as having nitrate concentrations approaching the drinking water standard of 10 mg/l nitrate-nitrogen (Holzmacher . . . 1970). A later study for Southold Township concluded: "It is perfectly clear that high nitrates in Southold Township are due primarily to fertilizers" (Holzmacher . . . 1972). Some typical nitrate results from wells in the area are shown in Table 1.

The hydrogeology of the farming areas (to be discussed later) limits the amount of water available. This, together with the fact that only expensive long-distance water transmission could

Table 1. Typical Nitrate (N) Concentrations in Eastern Long Island Ground Water

Well Location	*Date Sampled*	*NO_3-N (mg/l)*
Mattituck	10/22/75	16.3
Greenport	10/14/75	8.2
Southold	11/03/75	12.0
East Mattituck	09/16/75	13.1
Test Plot (Demarest)	03/26/75	23.0
Test Plot (Dickerson)	03/26/75	21.0
Test Plot (Halsey)	03/26/75	20.0
Test Plot (Babinski)	03/26/75	17.0

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alleviate the drinking water quality problem, indicates that to preserve and protect the eastern Long Island water resource the source of the nitrates, including fertilizers, must be limited or reduced.

Suffolk County ranks first in New York State in total agricultural sales. Potatoes, the principal field crop, accounts for about 25,000 acres (10145 ha) of the total of over 60,000 acres (24420 ha) in agricultural production. Other important agricultural activities include production of ducks, sod, nursery products, and a wide variety of fruits and vegetables (Suffolk County CES, 1973).

Potatoes and vegetable crops require intensive management, including relatively high rates of fertilization, for maximum production. In the past, the consequences of over-fertilization have been minor in contrast with yield losses associated with under-fertilization, and a trend of over-fertilization has developed. Environmental concerns and drastic increases in fertilizer prices have resulted in the need for a reappraisal of fertility management.

Results of a potato fertilization survey (Rykbost, 1976) indicate that application rates range from 125 to 310 lb-N/a (A) (139 to 344 kg-N/ha) with a median of 225 lb-N/a (250 kg-N/ha). Most of this is applied at planting. A significant portion of the N applied at planting is susceptible to leaching to the ground water prior to the period of maximum plant uptake. Nitrogen removal in the harvested tubers account for 75 to 150 lb-N/a (83 to 167 kg-N/ha) (Meisinger, 1976) depending primarily on crop yield.

Plant tops may contain up to 100 lb-N/a (111 kg-N/ha) at harvest, but this goes into the soil organic nitrogen pool which supplies approximately an equivalent amount to the next year's crop.

Since soil conditions are not favorable for denitrification or volatilization processes, it seems likely that significant quantities of nitrate-N may be leached to the aquifer under present fertilization practices.

Recharge of ground water on Long Island averages about 23 inches per year (58.4 cm). A loss of 50 lb-N/a (55.5 kg-N/ha) via leaching will be sufficient to maintain this amount of recharge at 10 mg/l NO_3-N, assuming no other nitrogen source. A change in potato fertilization practice is necessary to restore the eastern Long Island ground water to an acceptable level of nitrate. However, this must be accomplished without jeopardizing the potato industry.

Fertilization of home lawns and other turfgrass areas can also serve as a source of groundwater nitrates. A 1973 survey of fertilization practices for Long Island turfgrass indicated a range in fertilization rates from zero to about 350 lb-N/a (389 kg-N/ha) a year. While much of the N may be taken up by turf, N removal from the biosphere is nil, even in the case of removal of clippings since these are composted or used for mulch and eventually decay. Volatilization of N from turfgrass may be an important N sink; however, leaching of nitrates from heavily fertilized lawns is undoubtedly a source of ground-water contamination (Snow, 1976).

During the spring of 1973, the Suffolk County Department of Environmental Control (SCDEC), Cornell University, acting through the Vegetable Crops, Agricultural Engineering, and Agronomy Departments and the Long Island Vegetable Research Farm (VRS), and the Suffolk County Cooperative Extension Service (CES) began discussion on how the fertilizer question could be investigated. By April 1974, a contract was signed by SCDEC and Cornell University to perform a four-year study of N fertilization of potatoes and turfgrass.

The study was designed to formulate and evaluate alternative N fertilization schemes which would reduce nitrate leaching losses while maintaining potato yields and turfgrass quality. Initial stages of the study were to be conducted on the VRF facility. Promising alternative fertilization practices were to be demonstrated and evaluated on commercial potato fields and typical turfgrass areas. Finally, a public information program would be formulated by CES to encourage adoption of recommended practices.

The duration of the study is presently June 1978 which will allow for three full growing seasons at the demonstration sites. Final recommendations will be based on five full years of intensive research efforts. The importance of longevity in a study of this nature cannot be overemphasized.

The results thus far are of sufficient importance that the authors felt that an interim report should be presented to the scientific community before the entire study has been completed.

BACKGROUND

Long Island is the easternmost part of New York State and extends northeastward parallel to the continental coastline (Figure 1). It is bounded

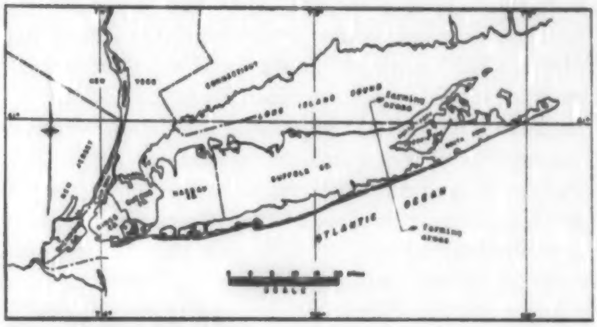

Fig. 1. Map of Long Island.

Fig. 2. Major landforms of Long Island, New York.

on the north by Long Island Sound, on the east and south by the Atlantic Ocean, and on the west by New York Bay and the East River. Politically, Long Island is divided into four counties: Kings, Queens, Nassau and Suffolk Counties. Kings and Queens Counties are part of New York City.

The topographical features of Long Island are the result of the glacial advance which occurred during the Pleistocene epoch that ended approximately 10,000 years ago. The major physiographic features (Figure 2) are: the terminal moraines that form the northern and central hills and "forks" of the island; the gently sloping outwash plain that extends southward from the hills; the deeply eroded headlands along the north shore; and the barrier beaches along the south shore. The total length of the island is 120 miles (193 km); its maximum width is about 23 miles (37 km); and its area is 1,400 square miles (3,626 sq km) (Suffolk County . . . , 1976).

The ground-water system of Long Island consists of five major geologic units (Figure 3). The uppermost formation is termed the Glacial aquifer consisting of sand and gravel of moderately high permeability and having an approximate maximum thickness of 400 feet (122 m). The Gardiner's clay is a formation which appears most extensively in the southern portion of the island and has an approximate maximum thickness of 150 feet (46 m). The Magothy aquifer is the largest formation consisting of coarse to fine sand of moderate permeability and an abundance of silt and clay of very low permeability. The approximate maximum thickness of this aquifer is 1,000 feet (305 m). Next is the Raritan clay which is a clay of relatively low permeability and an approximate maximum thickness of 300 feet (91 m). The final formation which rests on bedrock is the Lloyd aquifer consisting of sand and gravel of moderate permeability which may be as thick as 300 feet (91 m). These formations are saturated with water to an uppermost level which forms the water table (New York . . . 1968).

The water budget of Long Island may be generally represented by the equation:

$$R = P - (r + E)$$

where R = natural recharge to the aquifer,

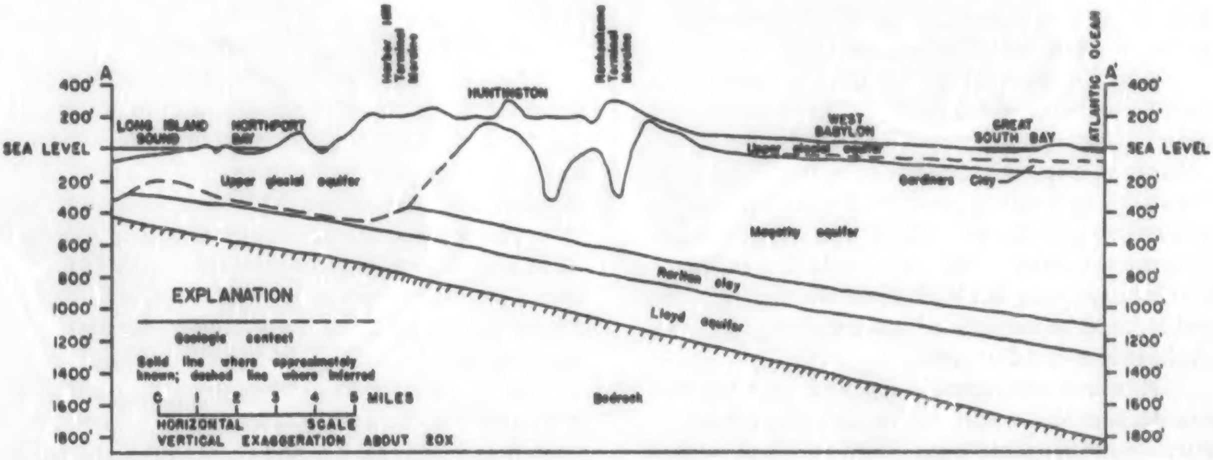

Fig. 3. Hydrogeologic cross section of Long Island.

P = precipitation,

r = direct runoff to streams and estuaries,

and E = evapotranspiration.

E is generally taken as 45 percent of P and r as 5 percent of P.

For Nassau and Suffolk Counties, ground water is the sole source of water and rainfall is the only replenishment.

Annual average precipitation is 44 inches (112 cm) (New York . . . 1968) with long-term high and low of 70 inches (178 cm) and 26 inches (66 cm) respectively (Suffolk County . . . , 1976).

The farming activity on Long Island is primarily centered in eastern Suffolk County and, more specifically, on the north and south forks. These two areas differ hydrologically from the main body of Long Island in that their aquifers are underlain by salt water. This limits the amount of water available for any type of use and causes pollution sources to be of greater concern.

The nitrogen budget for potato production involves a large number of input and output components. After an exhaustive analysis of this budget, Meisinger (1976) indicates that for Long Island the simple equation: NF = NT + NL can be used.

NF is N fertilizer applied in the crop year, NT is N removed in the harvested tubers, and NL is nitrate-N leached to the ground water. Since NT is nearly a linear function of crop yield (Meisinger, 1976), leaching losses will be determined by fertilizer N application rates and crop yields.

The above relationship suggests two alternatives for reducing leaching losses. The first and most obvious is to reduce N application rates. The second is to increase the nitrogen-use efficiency (NUE) by providing fertilizer N at the time it is needed and in a form which is available to the crop but not subject to leaching.

Applying most or all N at planting opens the door for substantial losses during major storm events in spring months. Under Long Island conditions, a split-application scheme should reduce spring leaching losses while maintaining or improving yields, provided the timing can be planned to coincide with crop needs. It was felt that by improving the NUE and reducing NF, the yields could be maintained and the nitrogen leaching loss could be reduced.

Research conducted at the VRF over the past two decades has shown that in most years most potato varieties achieve maximum yields at rates of 150 to 160 lb-N/a (167 to 178 kg-N/ha) (S. Dallyn, unpublished data). In years when excessive precipitation during the period from April through May leaches significant amounts of N past the crop root zone, yield responses to rates up to 200 lbs. N have been observed. Frequently, yield depressions have occurred in response to rates above 160 lb-N/a, due to delayed maturity and/or salt injury to seedlings. These data, coupled with grower practices identified (Rykbost, 1976) suggest that reducing N rates would not only reduce leaching losses but in many cases would result in increasing crop yields and dollar returns. Furthermore, crop quality may be improved as excessive N fertilization frequently results in an immature crop susceptible to bruising during harvest and decay in storage (Smith, 1968).

Leaching losses from N fertilization of turfgrass areas are difficult to quantify. Fertilization practices are varied and few records are kept. N usage by turf varies greatly depending on type of grass, mowing practices, irrigation practices, age of the turf, frequency, rate and timing of N application, and other factors. A 1973 survey of Long Island turf fertilization practices (Rykbost, unpublished data) indicated a range in N application rates from 0 to 350 lb/a (0 to 390 kg/ha).

Recent literature on turfgrass fertilization has been reviewed by Snow (1976), and a glaring lack of research relevant to the fate of N applied to turf is evident. Substantial volatilization N losses may occur from fertilizer as well as from decaying clippings. During turf establishment, N is rapidly incorporated into plant tissues and stable organic debris. However, after an initial period of biomass buildup, which may require 20 years or more, no obvious N sink can account for N additions. Thus, in established turf areas leaching losses from fertilization of turfgrass may outweigh the losses from fertilization of crop lands on Long Island.

POTATO EXPERIMENTS

Experiment I, conducted in 1972-1975, was designed to determine the timing of crop N uptake, and yield and NUE response to split applications. Results (Schippers and Rykbost, 1975) indicated that single N applications at planting were no better than split applications for crop yield except during seasons when heavy spring rains take place [11 inches (28 cm) of rain in June 1972], when substantial leaching losses occurred from single applications.

Split applications significantly increased tuber N removal for the farm year average. Weekly sampling of tops, tubers, and roots showed slow N uptake (30 lb-N/a — 33 kg-N/ha) until the 6-inch

Table 2. Total Yield and Tuber N Removal for Five Varieties and Four N Rates in 1974 and 1975

	N-Rate lb-N/a (kg-N/ha)		*Total Yield*[4] *100 lbs/a (kg/ha × 10^2)*		*Tuber N Removed*[4] *lb/a (kg/ha)*	
Variety	*1974*[1]	*1975*[2]	*1974*	*1975*	*1974*	*1975*
K	140 (155)	125 (139)	394 (437)^{bc}	363 (403)^{b}	112 (124)^{a}	120 (133)^{b}
C	140 (155)	125 (139)	454 (504)^{a}	415 (461)^{a}	118 (131)^{a}	142 (158)^{a}
H	140 (155)	125 (139)	400 (444)^{b}	374 (415)^{b}	108 (120)^{a}	120 (133)^{b}
S	140 (155)	125 (139)	374 (415)^{cd}	344 (382)^{b}	115 (128)^{a}	118 (131)^{b}
A	140 (155)	125 (139)	365 (405)^{d}	305 (339)^{c}	100 (111)^{a}	93 (103)^{c}
Avg.[3]	80 (89)	50 (56)	338 (375)^{c}	328 (364)^{b}	88 (98)^{b}	93 (103)^{c}
Avg.	120 (133)	100 (111)	398 (442)^{b}	355 (394)^{ab}	113 (125)^{a}	111 (123)^{b}
Avg.	160 (178)	150 (167)	427 (474)^{a}	380 (422)^{a}	120 (133)^{a}	132 (147)^{a}
Avg.	200 (222)	200 (222)	427 (474)^{a}	378 (420)^{a}	121 (134)^{a}	138 (153)^{a}

[1] Averaged over N rates of 80, 120, 160, and 200 lbs/a (89, 133, 178, and 222 kg-N/ha).

[2] Averaged over N rates of 50, 100, 150, and 200 lbs/a (56, 111, 167, and 222 kg-N/ha).

[3] Averaged over five varieties.

[4] Means followed by the same letter are not significantly different at the 5 percent level, based on the Duncan's multiple range test (Steel and Torre, 1960).

(15-cm) plant height stage. During the next three weeks, over 50 percent of total seasonal N uptake occurred. Thus, split applications must be carefully timed to insure that the N is available to the plant when needed.

Experiment II, conducted in 1973-75, compared N rates from 0 to 200 lb/a (0 to 223 kg/ha) single versus split applications and the nitrification inhibitor "N-Serve" [2-chloro-6 (trichloromethyl) pyridine] versus no inhibitor (Meisinger, 1976). A split N application with 50 lb/a (55.7 kg/ha) applied at planting and 100 lb/a (111 kg/ha) side dressed at the 6-inch (15.2-cm) height stage consistently achieved maximum yield. Split N applications significantly increased the total three-year tuber N yield, hence NUE. N-Serve delayed nitrification long enough to avoid early season leaching losses, increased total three-year tuber N yields, and increased residual soil N. Meisinger (1976) concluded that the split application at 150 lb-N/a (167 kg-N/ha) would produce maximum yields while maintaining leaching losses at acceptable levels.

Experiments I and II were conducted with the Katahdin potato variety which accounts for two-thirds of the Long Island crop. Other varieties will respond differently to a given fertilization program. Experiment III was conducted in 1974-1976 to evaluate five varieties at a range of N rates using split applications. Included were three full-season varieties: Katahdin (K), Cascade (C), and Hudson (H), and two early maturing varieties: Superior (S) and Alamo (A). In 1974, 40 lb-N/a (44 kg-N/ha) were applied at planting. At the 6-inch (15-cm) stage, side dress rates of 40, 80, 120 and 160 lb-N/a (44, 89, 133, 178 kg-N/ha) were applied.

In 1975, 50 lb-N/a (56 kg-N/ha) were applied at planting and side dress rates of 0, 50, 100 and 150 lb-N/a (0, 56, 111, and 167 kg-N/ha) were applied at the 6-inch (15-cm) stage. The interactions between variety and N rate were not statistically significant in either year for tuber yield or tuber N removal. Table 2 shows average yields and tuber N removals for both years. The most important conclusions to be drawn from these data are:

(1) Tuber yields were not increased in response to the last increment of N.

(2) Tuber N removal increased only slightly in response to the last two N increments, hence NUE decreased rapidly.

(3) Yield and NUE are variety dependent.

(4) Based on the equation NF = NT + NL, leaching losses exceeded 50 lb-N/a (55.5 kg-N/ha) in both years for the 200 lb-N/a (222 kg-N/ha) rate.

TURFGRASS EXPERIMENTS

A blend of four Kentucky Bluegrass varieties was seeded in May 1974 (Snow, 1976). Nitrogen rates of 0, 2, 4 and 8 lbs/1000 sq ft (0, .0098, .0195 and .039 kg/m^2) were applied in 0, 1, 2, 4 or 8 applications per year. Clippings were sampled weekly in 1975 to determine dry matter production and N removal. Biomass samples were taken in November 1974 and March, September and November 1975 to assess biomass N accumulation.

By November 1975, total N accumulated in the biomass where clippings were returned was 2.5, 3.3, 3.8, and 4.8 lbs/1000 sq ft (.0122, .0161, .0185, .0231 kg/m^2) for two-year total N applications of 0, 4, 8 and 16 lbs/1000 sq ft (0.0, .0195, .039, and .0781 kg/m^2) respectively. Where clippings were removed, biomass N was somewhat less, 2.2, 2.8, 3.6 and 4.0 lbs/1000 sq ft (.0107, .0136, .0175 and .0195 kg/m^2) respectively, for each application. These data indicate that a small percentage of fertilizer N is incorporated in plant material during the period of turf establishment. This percentage can be expected to diminish as the turf matures.

Snow (1976) concluded that:

(1) Where clippings are returned, N fertilization above a rate of 2 lbs/1000 sq ft (.0098 kg/m^2) may lead to ground-water contamination above 10 mg/l nitrate-N.

(2) Where clippings are removed, a large part of the applied N is removed from the system, but may eventually reach ground water. This is due to the fact that the clippings are normally composted by the homeowner or they decompose in the local landfill. The nitrogen still returns to the ground water.

(3) N recovery will decrease as the turf matures, and high rates of N on mature turf may lead to serious ground-water contamination.

(4) At low total N rates, more frequent applications will provide for more uniform growth and quality throughout the growing season.

(5) The use of low N-requiring varieties of turfgrass should be encouraged.

In September 1974, 18 Kentucky bluegrasses, 10 red fescues, 8 perennial ryegrasses, and 8 blended mixes were seeded under a minimum fertility maintenance schedule of 2 lb-N/1000 sq ft (.0098 kg-N/m^2) applied in ½ lb (.22 kg) increments in September and October 1974, and April and May 1975. The replicated plots are under routine observation to evaluate turf performance and isolate those varieties which can maintain acceptable visual quality. A minimum of three years of evaluation will be needed to select low maintenance turfgrass. As expected, a wide range in quality response has been observed to date, with several varieties performing very well.

DEMONSTRATION PLOTS

The objectives of the potato fertilization demonstration trials were to:

(1) Verify results from research conducted at the Research Farm on commercial growers' fields;

(2) Demonstrate alternative fertilization practices which should result in equivalent yields, reduced costs for fertilizer, and reduced levels of nitrate nitrogen in ground waters underlying agricultural fields;

(3) Monitor the impact of nitrogen fertilization practices on ground water; and

(4) Encourage growers to adopt new practices.

Four demonstration sites were selected in January 1975. Growers involved agreed to conduct the trials for at least two years. Shallow skimming wells (two wells per site) were installed to allow monitoring of ground-water nitrogen under "grower practice" and "recommended practice" treatments.

The demonstration plots were divided in half with the "recommended practice" of 1,670 lb/a (186 kg/ha) of 3-18-9 fertilizer and 300 lb/a (334 kg/ha) of aeroprills (34-0-0) applied at the 4-6-inch (10-15.2-cm) plant height, side dressed in bands. The "grower practice" varied from farm to farm (see Table 3).

The skimming wells were all suction well, 2-inch (5.08-cm) diameter, with a 5-foot (1.52-m) stainless steel well screen. Top of screen was set 1 foot (.304 m) below the water table. Initial samples were taken before planting and two-week sampling intervals were followed thereafter. Samples were collected after pumping approximately 50

Table 3. Demonstration Plot Information, 1975

Plot No.	*Location*	*Size a (ha)*	*Soil Type*	*Depth to Water ft (m)*	*Elev. of GWT at MSL, 3/76 ft (m)*	*Recommended N-Rate lbs/a (kg/ha)*	*Grower N-Rate lbs/a (kg/ha)*
1	North Fork	2.0 (.81)	Haven	3.3 (1.0)	0.7 (.2)	133 (148)	200 (224)
2	North Fork	1.6 (.64)	Haven	16.5 (5.0)	3.9 (1.18)	154 (172)	217 (243)
3	South Fork	3.5 (1.41)	Bridgehampton	11.0 (3.3)	4.0 (1.21)	149 (167)	270 (302)
4	South Fork	2.5 (1.01)	Plymouth	4.1 (1.3)	4.4 (1.34)	148 (166)	185 (207)

Table 4. Demonstration Plot Results

		Applied N lbs/a (kg/ha)	*Yield Cwt/a (kg/ha)*	*Initial Nitrate-N (mg/l) 3/20/75*	*Mid-Sampling NO_3-N (mg/l) 10/3/75*	*1-Year Sampling NO_3-N (mg/l) 3/15/76*
Site 1– Orient Point	G.P.[1]	200 (223)	384 (42800)	16.3	20	14.2
	R.P.[2]	133 (148)	323 (36000)	25.0	22.8	18.9
Site 2– Southold	G.P.	217 (241)	259 (28900)	21.8	16.5	16.4
	R.P.	154 (171)	256 (28530)	20.8	19.2	21.0
Site 3– Wainscott	G.P.	270 (301)	259 (28900)	8.3	6.2	11.9
	R.P.	149 (166)	225 (25000)	8.0	12.3	5.9
Site 4– Water Mill	G.P.	185 (206)	290 (32300)	23.4	20.0	13.5
	R.P.	148 (164)	313 (34900)	24.9	24.0	14.1

[1] Grower Practice.

[2] Recommended Practice.

gallons (189 liters) and a micro-Kjeldahl steam distillation procedure was used to determine ammonium, nitrite and nitrate nitrogen.

Throughout the well sampling program, ammonium and nitrite nitrogen levels in all wells have been negligible in comparison to nitrate nitrogen contents. Ground-water nitrogen in the discussions will refer only to nitrate nitrogen.

The results from Site 1 are shown in Table 4. Only 75% of the recommended second N application was actually applied, and the method application used was top dressing as opposed to the more efficient side dressing method. These factors were important in contributing to a lower potato yield on the r.p. treatment.

The ground-water nitrate concentrations are plotted in Figure 4. The one-year's sampling shows a decrease in ground-water nitrate on both plots, but the recommended rate shows a greater decrease.

Site 2 was irrigated three times during the growing season and no problems were encountered. The yields were equivalent with a 63 lb-N/a (70 kg-N/ha) fertilizer reduction.

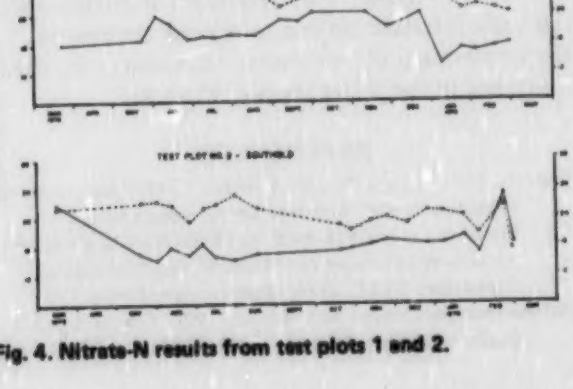

Fig. 4. Nitrate-N results from test plots 1 and 2.

The ground-water nitrate concentrations are plotted in Figure 4. They show the ground-water quality in the growers' field to be consistently lower throughout the season (see comments further on).

Site 3 was not irrigated during the growing season, and the fact that the "recommended practice" area did not receive the same weed killing treatment as the "grower practice" area contributed to the yield differential. Figure 5 shows a ground-water improvement in the "recommended practice" area after one year of operation.

At Site 4 the yields were satisfactory and serve to demonstrate that the reduced fertilizer applications can produce equivalent and even greater crop yield. Nitrogen levels in both areas decreased on both treatments (see Figure 5).

Figures 4 and 5 present graphs of the nitrate nitrogen concentrations by site number, treatment, and sampling date. Wide variations were observed between sites, between treatments on a given site, and between sampling periods for a given well. Some generalizations can be made from the data.

(1) At each site, nitrate nitrogen contents in excess of 10 ppm (and frequently more than double or triple that value) were observed.

(2) Since these fields all have an agricultural history, the major source of ground-water nitrogen is fertilizer.

(3) As the season progressed, differences in nitrate levels between treatments on a given site gradually decreased. On June 12, 1975, the average nitrate concentration was 9.0 ppm higher on the "recommended practice" treatment. This difference

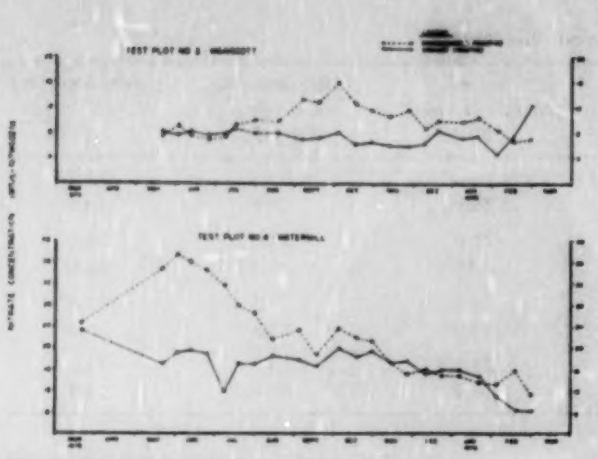

Fig. 5. Nitrate-N results from test plots 3 and 4.

gradually dropped and on March 1, 1976, levels were the same on both treatments.

(4) A statistically significant trend towards lowering nitrate levels on the "recommended practice" treatment can be seen in Figure 6. However, an additional year's data will be needed to confirm this apparent trend (VRF, 1976).

(5) A period of time is needed to allow for soil and ground-water reaction and adjustment. This time period appears to be at least a growing season.

A review of the 1975 ground-water nitrogen data has resulted in the expansion of the ground-water sampling network at each site from two wells to six wells per site. In addition, detailed ground-water directions were obtained and soil analyses performed. Data collected for the 1976 growing season has shown that the nitrate levels on all four fields is lower in the "recommended practice" than the "grower practice." Additionally, at two of the sites, the quality of the ground water is below 10 mg/l nitrate-N standard (authors' observation of 1976 data).

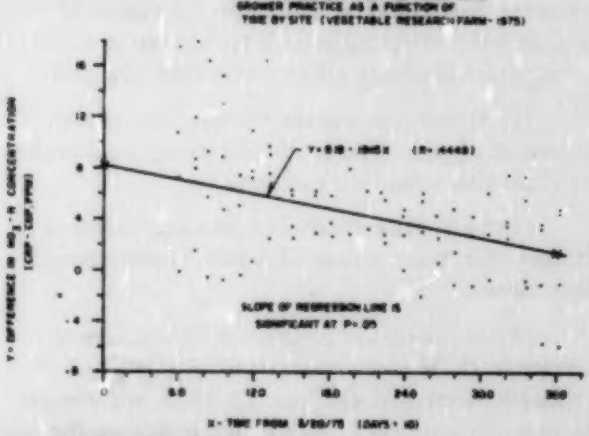

Fig. 6. Nitrate-N regression line.

CONCLUSIONS

The following represent the important results thus far:

(1) NUE decreases rapidly when fertilizer rates exceed 150 lb-N/a (167 kg-N/ha).

(2) Two fertilizer applications—one-third at planting, two-thirds at the 4- to 6-inch (10-15.2-cm) plant height—will result in maximum NUE and still produce economical yields.

(3) Reducing fertilizer application to 150 lb-N/a (167 kg-N/ha) with split applications appears to result in equivalent crop yields, optimum NUE and a reduction in nitrogen leaching, to a point which will maintain ground-water quality below 10 mg/l-nitrate-N.

(4) Nitrogen/fertilizer is a major source of nitrate in ground water beneath agricultural areas.

(5) Immediate ground-water response to changes in nitrogen applied to the soil cannot be expected.

(6) The ground-water quality is changing in direct relation to the amount of fertilizer applied.

(7) N fertilization of turfgrass has the potential of contributing more to the ground-water deterioration than fertilization of potatoes.

If the study continues providing successful crop yields at low fertilization rates and the ground-water quality continues to show improvement, a very serious ground-water pollution problem will be solved and restoration of the aquifer's quality can begin.

ACKNOWLEDGEMENTS

Thanks to Dr. A. Schippers for statistical and scientific information and to various members of the Department of Environmental Control for their assistance in the preparation of the text.

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DISCUSSION

The following questions were answered by Joseph H. Baier after delivering his talk entitled "The Contribution of Fertilizer to the Ground Water of Long Island."

Q. by K. Childs. *Your presented results suggest that lower nitrate concentrations in the ground water correspond to higher application rates—why?*
A. A certain period of adjustment appears to be taking place within the soil mantle before nitrogen equilibrium is reached and the effects of reduced fertilizer application can be shown by the ground water. 1976 QW data indicate that nitrate in ground water under recommended fertilizer plots is lower than that of grower practice. Finally, we have no historical data from the demonstration sites that could help us review the seasonal fluctuations.

Q. by Lawrence C. Eccles. *You assume that nitrogen not used by the crop is lost to the ground water. Did you evaluate losses by denitrification also?*
A. The Long Island outwash plain sands and gravels are of such a nature that solubilized nitrates once through the root zone pass directly to the ground water without change. The entire nitrogen balance (input and output) is summarized by Meisinger in his Ph.D. thesis at Cornell University, "Improving the Efficiency of Fertilizer Use for Potatoes."

Q. by Charles P. Vanderlyn. *How were you able to separate the source of the nitrogen in the ground water between that applied as fertilizer versus the ammonia leached from the proliferation of septic tanks on Long Island? What was the difference between N readings in surface water and ground water?*
A. The area is rural, and the only septic tank influence would be the occasional farm house. However, wells were put upgradient of the farms and sample results have shown the nitrogen concentration to be extremely low.

In the areas where the farming communities lie, there are virtually no surface-water bodies which would receive runoff.

Q. by Mike Apgar. *Are the wells located on, or downgradient from the test plots? How far?*
A. There are no wells downgradient of the test plots—only upgradient. The upgradient wells were immediately adjacent to the sites.

Q. by Charles Kreitler. *What are the different nitrogen forms predominantly used in the fertilizer in Suffolk County? Is denitrification or ammonia volatilization occurring?*
A. The main nitrogen form of fertilizer is nitrate. Some, but very little amounts of urea are used. Even less are the amounts of inhibitors used.

Denitrification and volatilization may be occurring; but in the over-all nitrogen budget, they amount to very small losses. Meisinger in his thesis at Cornell University ("Improving the Efficiency of Fertilizer Use for Potatoes") has presented an exhaustive nitrogen budget analysis.

Q. by Abe Kreitman. *How is rainfall taken into account (or supplementary irrigation) subsequent to fertilizer application, and what were the antecedent conditions?*
A. Little to no irrigation was experienced on four plots. This can be said of the Long Island potato farming in general. Heavy spring rains often play a part in the fertiliza-

tion practice of the farmers in that they will go back a second time and apply more fertilizers if they feel that spring rains have washed away the initial application.

Q. by Jeffrey Gilman. *Can you attribute higher NO_3 concentrations in your recommended practice to your recommended method of application (side dressing)?*
A. Side dressing fertilization allows the fertilizer to be placed almost directly on top of the planted seed where it can be used immediately. Side dressing is used by some farmers, but no study was ever conducted to try to compare the yields of side dressing versus top dressing.

Q. by P. O. Seman. *Did you consider nitrification in your budget to get the input to ground water? If not, could you estimate the error?*
A. Yes, the nitrification of organic material from plowed-under plants is a consistent nitrogen source throughout the year. This has been quantified as approximately 100 lbs/acre/year.

Q. by N. W. Johnson. *What was the form of the nitrogen fertilizer applied? Was volatilization considered in your mass balance?*
A. Fertilizer applied was in the nitrate form. Second, the mass balance did volatilize but it was negligible in comparison to the quantity of fertilizers being applied and/or lost to the ground water.

Q. *Has anyone done a historical land-use study on Long Island in relationship to all agricultural practices and ground-water quality?*
A. This has been done in piecemeal fashion, and it is presently being redone in all agricultural practices. Feedlot and animal wastes are being quantified under the Federal 208 study.

Q. *What assurance do you have that the sampling done in those years represent the different practices?*
A. The areas used in the demonstration plots have all been farmed for many years before the demonstration plots were used. By reviewing the amounts of fertilizer applied by the growers, we have ascertained that these are average applications.

Abatement of Nitrate Pollution in a Public-Supply Well by Analysis of Hydrologic Characteristics[a]

by Lawrence A. Eccles, John M. Klein, and William F. Hardt[b]

ABSTRACT

High concentrations of dissolved nitrate in the ground water in the Redlands, California, vicinity threaten public water supplies. Dissolved nitrate-nitrogen concentrations in water from wells, which frequently exceed 10 mg/l (milligrams per litre), are attributed to the previous applications of large quantities of commercial nitrogen fertilizer on citrus crops. In a city of Redlands public water-supply well field, wells 1 and 2 normally produce water with dissolved nitrate-nitrogen concentrations of about 18 and 30 mg/l, respectively. Well 1 is a large-capacity well capable of yielding 3,700 gal/min (gallons per minute) [233 l/s (litres per second)] and is the major source of water in the well field.

The very permeable unconfined alluvial aquifer is composed of sand, gravel, boulders, and discontinuous clayey deposits. Well 1 is 742 feet (226 metres) deep and is perforated throughout most of the zone of saturation. A major clayey interval from 425 to 480 feet (130 to 146 metres) effectively separates the aquifer into an upper and lower zone. Well 2 is 426 feet (130 metres) deep and is perforated throughout most of the upper zone of saturation. At the well field the static water level (February 1976) was 180 feet (55 metres) below land surface.

Independent tests were made on wells 1 and 2 to evaluate aquifer characteristics and to determine the sources of the high-nitrate water. Chemical analyses of water collected from well 1 during the first 48-hour test showed an increase in dissolved nitrate-nitrogen from 4.1 mg/l to a maximum of almost 20 mg/l. Dissolved nitrate-nitrogen concentrations during the 48-hour test of well 2 stayed constant at 30 mg/l. Interpretation of the chemical data and the results of previous studies indicate that the upper zone of saturation is the higher in concentration of dissolved nitrate.

To reduce the concentration of dissolved nitrate in water from well 1, an inflatable packer was placed in the casing at 480 feet (146 metres) to coincide with the bottom of the clayey interval. The packer sealed off the upper part of the well and, as determined from a final test of well 1, reduced dissolved nitrate-nitrogen concentrations from 20 to 4 mg/l while only reducing well yield from 3,700 to 2,600 gal/min (233 to 164 l/s).

INTRODUCTION

Redlands, California (Figure 1), is in a semiarid area in which nearly a century of irrigated citrus culture has contributed to dissolved-nitrate concentrations in water from wells frequently exceeding the recommended nitrate criterion for public water supplies of 10 mg/l (milligrams per litre) nitrate as nitrogen (Environmental Protection Agency, 1972). The citrus groves (Figure 2) are gradually being replaced by urban development resulting in increased use of ground water for

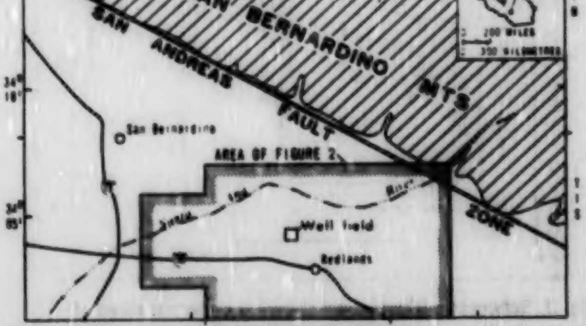

Fig. 1. Index map of Redlands and vicinity.

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

[b] Hydrologists, U.S. Geological Survey, Laguna Federal Building, 24000 Avila Road, Laguna Niguel, California 92677.

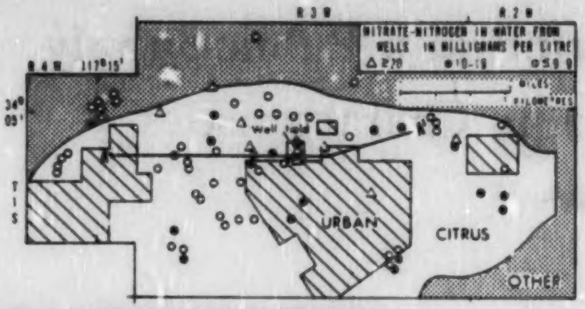
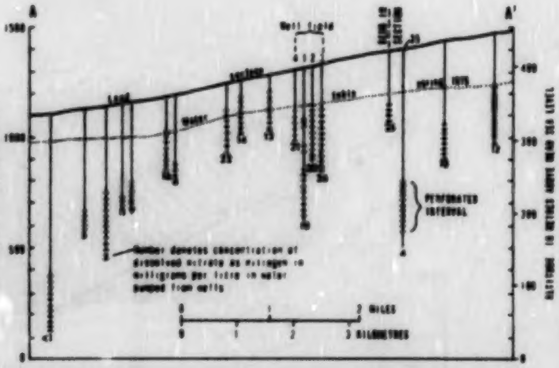


Fig. 2. Land use and nitrate distribution.

domestic and industrial purposes. The California Department of Health requires that remedial action be taken to meet the 10 mg/l criterion for public water supplies. In the city of Redlands Texas Street well field (Figure 3), wells 1, 2, and 3 produce water with dissolved nitrate as nitrogen concentrations of 18, 30, and 26 mg/l, respectively. Well 1 is the largest capacity well and the major source of water in the well field.

As part of an ongoing investigation in cooperation with the San Bernardino Valley Municipal Water District, the California Department of Health, and the city of Redlands, the U.S. Geological Survey is making an appraisal of the hydrogeologic system to aid in an attempt to reduce the concentration of dissolved nitrate in well 1. The objectives of the cooperative investigation summarized herein were to determine the hydrologic characteristics of the aquifer near the well field in order to suggest certain modifications to the construction of well 1 that might satisfactorily reduce the dissolved-nitrate concentration in the water while minimizing the reduction in well yield.

In this report, nitrate concentrations are expressed in terms of an equivalent amount of elemental nitrogen ($(NO_3) = 4.43\ (N)$).


A
A'
1500
1000
500
0
400
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200
100
0
ALTITUDE, IN FEET ABOVE MEAN SEA LEVEL
ALTITUDE, IN METRES ABOVE MEAN SEA LEVEL
Well field
Land
surface
Water
table
spring 1975
PERFORATED
INTERVAL
Number denotes concentration of dissolved nitrate as nitrogen in milligrams per litre in water pumped from wells
<1
0
1
2 MILES
0
1
2
3 KILOMETRES

Fig. 3. Schematic illustration showing concentration of dissolved nitrate and perforated interval in wells along A-A'.

HYDROGEOLOGY

Redlands is in the upper Santa Ana River basin at the east end of the San Bernardino Valley near the base of the San Bernardino Mountains (Figure 1). The valley is filled with alluvium consisting of sand, gravel, and boulders interspersed with lenticular deposits of clay and silt. The maximum depth to bedrock is about 1,000 feet (305 metres) (Burnham and Dutcher, 1960; Dutcher and Garrett, 1963).

The alluvial aquifer within the study area is considered to be unconfined, although discontinuous intervals of clay can cause short-term semiconfined conditions. The depth to water below land surface in the Redlands vicinity ranges from 20 to 300 feet (6.1 to 91 metres). At the Texas Street well field the static water level was 180 feet (55 metres) below land surface in February 1976. The general gradient for both land surface and ground-water surface is from east to west (Figure 3). Ground water is replenished from natural and imported flow in the Santa Ana River, artificial recharge, irrigation return, and direct infiltration of precipitation.

As evidenced by Figure 2, no clearly defined pattern for the areal distribution of dissolved nitrate in ground water exists in the Redlands area. A study by L. A. Eccles and W. L. Bradford (U.S. Geological Survey) in 1975 gave some indication of the areal and vertical distribution of dissolved nitrate so that a conceptual model could be designed. Their findings showed that some differences in dissolved-nitrate concentrations (Figure 3) correlated with (1) the different depths and perforated intervals of the wells, (2) the location of semiconfining clay intervals, and (3) to some extent the depth of the pump intake.

Well 1 (Figure 4) was drilled to 742 feet (226 metres) and is perforated from 240 to 700 feet (73 to 213 metres) below land surface, with the exception of two zones unperforated opposite

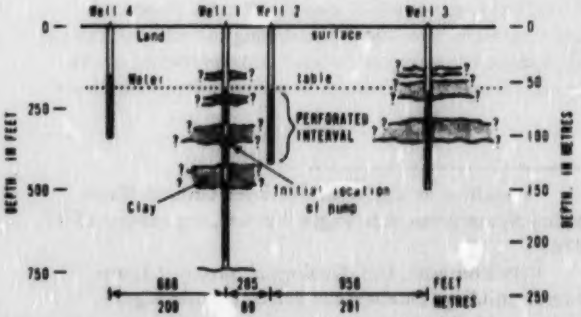

Fig. 4. Schematic illustration of well field.

sandy-clay intervals at 302-360 and 423-490 feet (92-110 and 129-149 metres). Well 1 contains two significant zones: the upper zone of saturation from 240 to 425 feet (73 to 130 metres) and a lower zone, below the lowermost clay interval from 480 to 700 feet (146 to 213 metres).

Well 2 is 285 feet (87 metres) from well 1 and 426 feet (130 metres) deep, with perforations throughout the upper saturated zone. The upper sandy-clay intervals encountered in well 1 were not reported in well 2, although the bottom of the well coincides with the top of the lower sandy-clay interval. Well 3, which is 956 feet (291 metres) from well 2, was drilled to 500 feet (152 metres) and perforated from 200 to 482 feet (61 to 147 metres), although the zone from 285 to 355 feet (87 to 108 metres) was reported by the driller to contain a significant clayey fraction.

In addition to the public water-supply wells in the well-field vicinity, several nearby large irrigation wells (Figures 2 and 3) pump water containing dissolved-nitrate concentrations of as much as 25 mg/l nitrate as nitrogen.

Nearby public-supply well 35, approximately 1 mile (1.6 kilometres) northeast of the Texas Street well field (Figure 3), is pertinent to this study as it was drilled to 926 feet (282 metres) and is perforated from 500 to 809 feet (152 to 247 metres), or only in the lower part of the aquifer. The dissolved nitrate in water from this well is generally about 4 mg/l nitrate as nitrogen, which is probably indicative of nitrate concentrations in the lower part of the aquifer.

A preliminary mass-balance calculation indicated that the upper zone contributes 60 percent of the water and 90 percent of the dissolved nitrate while the lower zone contributes 40 percent of the water and only 10 percent of the dissolved nitrate in well 1. It was expected that after modification, well 1 would produce water from the lower zone containing dissolved nitrate similar to the concentration found in the water from well 35.

APPLICATION OF AQUIFER TESTS TO NITRATE ABATEMENT

An analysis of the hydrologic characteristics of the alluvial aquifer coupled with the hydraulics of the public-supply wells for the city of Redlands is essential in determining the behavioral pattern and possible means for abatement of ground-water nitrate pollution. Controlled tests in which wells are pumped to note aquifer response and changes in chemistry of the water pumped from the well are one method to help develop a conceptual model of the hydrochemical flow system in the aquifer.

Two independent tests (1976) were made in wells 1 and 2 to evaluate aquifer characteristics, changes in water quality, and sources of high-nitrate water. A third and final test was performed in well 1 after it had been modified, on the basis of interpretation of the results of the two prior tests.

The first test consisted of pumping well 1 for 48 hours at a rate of 3,700 gal/min (gallons per minute) or 233 l/s (litres per second) with a drawdown of nearly 17 feet (5.2 metres). Specific capacity was computed to be 220 gal/min per foot (45 l/s per metre) of drawdown. Water-level measurements were also made in nearby observation wells 2, 3, and 4. The pump was set at 360 feet (110 metres) with all the water entering the pump column at the end of the suction pipe at 392 feet (119 metres) below land surface. Chemical analyses of water collected from well 1 during the test showed an increase in dissolved nitrate as nitrogen from 4.1 mg/l to a maximum of almost 20 mg/l (Figure 5a).

Because of a 274-foot (84-metre) difference in the depths of wells 1 and 2 and the effects of partial penetration in analyzing aquifer transmissivity by the Theis and Jacob methods, a second test was conducted by pumping the shallower well 2 to better define the upper zone. This well was pumped for 48 hours at a rate of 2,270 gal/min (143 l/s) with a drawdown of about 13 feet (4 metres). Specific capacity was computed to be 174 gal/min per foot (36 l/s per metre) of drawdown. Water-level measurements were made in nearby observation wells 1 and 3. Dissolved nitrate as nitrogen in water from well 2 stayed constant at 30 mg/l during the test (Figure 5b). This would indicate that the upper part of the aquifer as defined by the depth of well 2 is high in dissolved nitrate.

Results of the two tests from the two wells of different depths indicate a range in aquifer transmissivity that can be largely related to differences between the zones in the aquifer. Transmissivity of the upper aquifer zone was computed to be 40,000-50,000 feet squared per day (3,720-4,640 metres squared per day).

Aquifer transmissivity as determined by pumping well 1 indicated values of 64,000-94,000 feet squared per day (5,950-8,730 metres squared per day). These values include both the upper and lower saturated zones. By subtracting transmissivity values determined by test 2 from test 1, values of

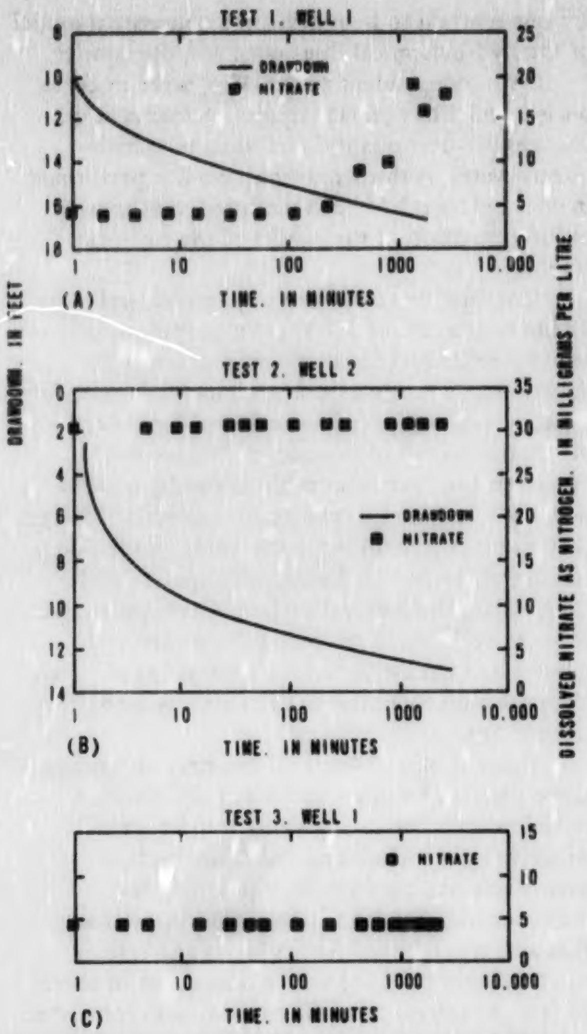

Fig. 5. Drawdown and dissolved nitrate (a) test 1, well 1, (b) test 2, well 2, (c) dissolved nitrate, test 3, well 1.

24,000-54,000 feet squared per day (2,230-5,020 metres squared per day) were computed for the lower zone. Thus, aquifer transmissivity values indicate a high-yielding lower zone. These values compare favorably with data from the nearby public-supply well 35, which shows a yield of 1,417 gal/min (89 l/s) with 12 feet (3.7 metres) of drawdown, for a specific capacity of 118 gal/min per foot (24 l/s per metre) of drawdown. This translates into an aquifer transmissivity of about 31,500 feet squared per day (2,930 metres squared per day).

Data analyses indicate a hydraulic conductivity of the aquifer, in the vicinity of the pumped wells, of about 115 feet per day (35 metres per day), with the lower zone somewhat less permeable than the upper zones. All test results indicate semi-artesian conditions, but the long-term effects indicate water-table conditions.

Interpretation of data from the two tests and of data from well 35 was used to design a third test to determine if the high-nitrate problem in well 1 could be alleviated. An inflatable packer was installed in well 1 at 480 feet (146 metres), at the base of the lower sandy-clay interval reported in the driller's log and confirmed by a natural gamma-radiation log. The pump was reset at 500 feet (152 metres) with the water intake at 532 feet (162 metres) below land surface (Figure 6). Well 1 was then pumped at a rate of 2,600 gal/min (164 l/s) for 48 hours. Water-level response was not measured below the packer. Throughout the test, the dissolved nitrate as nitrogen concentration remained at 4 mg/l (Figure 5c), the same as in the water from well 35. Thus, as a result of the well modifications, nitrate concentration decreased about 80 percent while the well yield decreased only about 25 percent, in comparison with the yield from the full aquifer thickness tested in the first test.

The water level in the well, representing aquifer head in the upper zone, declined about 2 feet (0.6 metres) during the third test, although the total water-level decline in the well must have been much greater, based on water-level measurements from the previous tests. This minimal decline indicates the lower sandy-clay interval and the packer in the well retarded the downward movement of high-nitrate water from the upper zones of the aquifer. The results of this third test suggest that permanent modifications to well 1 necessary to seal off the perforations above 480 feet (146

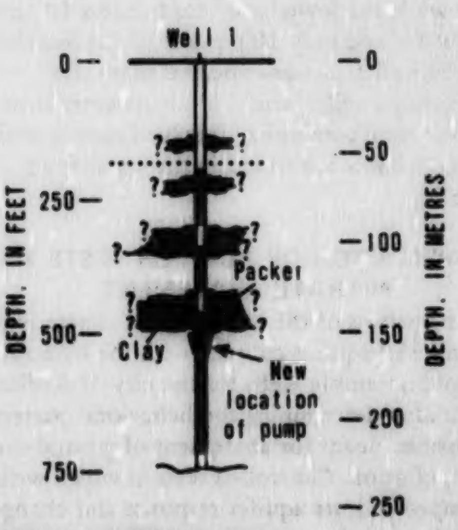

Fig. 6. Well 1 as modified for test 3.

metres) would be useful to satisfactorily reduce the dissolved-nitrate concentrations in the water while maintaining an acceptable well yield.

CONCLUSION

Results of this evaluation and the ensuing solution for reducing the dissolved nitrate in water pumped from wells has applicability throughout the Redlands and other hydrologically similar areas where there has been heavy nitrogen fertilization with resultant pollution of ground water used for a public supply.

Present practices of indiscriminant pumping from various producing zones in an aquifer and from wells of different depths and perforated intervals can result in mixing, throughout a well bore, native water containing low concentrations of dissolved nitrate with water containing high concentrations of dissolved nitrate derived from surface sources. Modification of wells to limit public-supply production to the lower parts of the aquifer, particularly below clay beds, may result in temporary abatement of the high concentrations of dissolved nitrate in water pumped from public-supply wells, but long-term abatement of dissolved nitrate in ground water would require a more sophisticated management program. Such a long-term abatement procedure could be accomplished by (1) minimizing the downward movement of water in an aquifer and, thus, avoiding the progressive deeper mixing of the polluted water with the native water; (2) avoiding the indiscriminant perforating of wells in all water-bearing zones penetrated; and (3) pumping several wells at lower rates rather than a single well or a few wells at maximum rates.

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DISCUSSION

The following questions were answered by Lawrence A. Eccles after delivering his talk entitled "Abatement of Nitrate Pollution in a Public-Supply Well by Analysis of Hydrologic Characteristics."

Q. *Was the preliminary mass balance calculation and the indirectly calculated transmissivity value for the lower part of the aquifer confirmed by other methods?*

A. The purpose of the preliminary mass balance calculation for the water produced from well 1 was to determine if there was significant production from the lower part of the aquifer. Additional development of the lower zone in well 1 occurred when the pump was relocated and packer installed. Direct calculation for a transmissivity value for the lower part of the aquifer was not made because the device intended to measure drawdown below the packer failed. Transmissivity for the lower zone of well 1 compares favorably with the estimated transmissivity for well 35.

Q. *Was there vertical leakage and if so what are the effects of vertical leakage in the aquifer?*

A. Results of test 3 indicated vertical leakage during pumping. Vertical leakage may also occur in unused wells that are perforated in both upper and lower parts of the aquifer. The upper "sandy-clay" intervals reported in the drillers' logs for wells 1 and 3 were probably mostly sand. That would explain why no "sandy-clay" intervals were reported in well 2 and why well 3 was perforated in a sandy-clay interval. The vertical leakage through the upper part of the aquifer was expected and nitrate data from test 2 indicated that it had occurred. The change in nitrate concentrations during test 1 probably represents stabilization of the distribution of production between upper and lower zones and an increase in the rate of leakage induced by pumping the well. Vertical leakage can be reduced by pumping several wells at lower rates rather than a single well or fewer wells at higher rates.

Q. *How long will the solution last?*

A. The solution could be long-term depending on management of withdrawal and recharge in the basin. Irrigation return is only one of several sources of recharge. Future prospects are for less irrigation return in comparison to artificial recharge and for incentives to water producers to withdraw irrigation supplies from the upper part of the aquifer and domestic supplies from the lower part.

Q. ***How much did this study cost?***
A. This hydrochemical study of a local well field was not a separately conceived or funded project, but rather a small part of other regional nitrate pollution and flow digital model studies. These interrelated studies kept costs at a minimum. This test presented an opportunity to obtain badly needed aquifer characteristics of transmissivity and storage, and time dependent nitrate analyses in the Redlands area. In addition, it was helpful to the cooperator in alleviating a high nitrate problem in a public-supply well. The cost of installing the packer and relocating the pump was $3,000 and chemical analysis for major constituents in water about $600. The cost of the pump or aquifer test including interpretation and administration was estimated to be about $3,000. Thus, the total cost is estimated to be about $6,600. Without the previously mentioned regional interrelated studies as a source of background knowledge, costs could conceivably have been much higher.

Q. ***On the final test with the packer, did you monitor water levels in the observation wells and if so, did the data confirm "T" computed from previous tests?***
A. Yes, the water levels were measured in observation wells 2 and 3 during the final test with the packer installed in well 1. The water-level declines in well 2 were dampened by the sandy-clay interval located above the packer and pump in well 1. Well 3 is perforated to 482 feet and water-level declines were similar to declines in test 1, prorated to the difference in well yield between tests 1 and 3. Water levels were not measured below the packer in the pumped well. An air line was set below the packer, but it leaked. Supporting data from nearby well 35 (perforated in the lower part of the aquifer only) indicated the previously determined transmissivity values were reasonable.

Design and Optimization of Ground-Water Monitoring Networks for Pollution Studies[a]

by H. O. Pfannkuch[b] and B. A. Labno[c]

ABSTRACT

The successful design and operation of a ground-water monitoring and surveillance system are based on a stepwide process of obtaining hydrogeologic information.

Because of the inherent uncertainty and inhomogeneity of natural hydrogeological systems, the true monitoring network cannot be specified before some basic knowledge about system configuration and dynamics is known.

Forethought, planning and incorporation of design criteria as part of the initial phase of project management establishes the monitoring network strategy. Optimization of the monitoring network takes place through the completion of the following five phases: (1) preliminary network design and information gathering, (2) initial installation and testing, (3) completion and verification, (4) operational, and (5) project termination.

Experience gained from monitoring the conditions at the University of Minnesota's chemical and special waste disposal site resulted in the design and optimization procedure. Concern for possible ground-water contamination led to analysis of surface and subsurface, physical and chemical conditions. Subsequently a monitoring system was established to meet project objectives. No degradation of the ground water was found during the five-year study.

INTRODUCTION

For any type of ground-water management operation, design and optimization of a ground-water monitoring network is vitally important to the success of the project. Since project objectives and the general nature of the study area will vary from project to project, forethought, planning and incorporation of design criteria at the beginning are essential to limiting additional costs and problems later.

The successful design and operation of a ground-water monitoring network are based on a stepwise process of obtaining hydrogeologic information. This, in turn, is used as input for a more rational design of the network to meet project objectives. The functions of the ground-water monitoring network are threefold. First, it needs to establish the basic structure, geometry and hydrologic characteristics of the flow field. Second, it needs to establish the dynamic reaction of the system under natural and stressed conditions. It is only after these two steps have been determined, and after a decision has been made of what critical quantities to monitor, that the third step, the final network density and configuration, can be defined.

Because of the inherent uncertainty and inhomogeneity of natural hydrogeological systems, the true monitoring network cannot be specified before some basic knowledge about system configuration and dynamics is known. Under the usual budgetary constraints, an optimum design for an observation network would be one where network points obtained in the first two steps can be efficiently used in the over-all and final configuration.

Ground-water pollution studies in general deal with problems of identifying pollution sources and containment of spreading from known sources, or elimination of the stress conditions. The other class of problems seeks to protect ground-water resources from pollution or from adverse effects due to overuse. Due to the nature of hydrogeologic systems, basic information is usually scarce and cost for new information is relatively high. The changing nature of geologic materials and their inherent inhomogeneities raise questions about network density and representativity of monitor samples.

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These difficulties should underline the importance of proper and efficient design of monitor networks, since every sample and every point have to count by holding maximum information content.

The purpose and scope of this paper is to provide a step-by-step thought process for developing a monitor network. It defines and describes five successive phases of the monitoring process. The information gained through monitoring in one phase serves as basis for the rational design of the next phase. The general principles are illustrated and reinforced by an actual example of a monitoring design for a waste disposal operation, and experience gained from monitoring the conditions at the University of Minnesota's chemical and special waste disposal site.

MANAGEMENT CONSIDERATIONS AND OBJECTIVES

Before developing the principles of monitoring system design, it is necessary to show how the information gathering and monitoring process interact with and depend on the over-all management scheme. The information gathering and the monitoring processes are an integral part of management because they enter into the decision-making process. In this context, information has value since increased information allows more efficient management, and it tends to reduce risk (hence loss) when a choice has to be made between several alternatives (Pfannkuch, 1975).

Because of this interrelation and interdependence, two sets of goals or objectives can be defined. One is the set of objectives formulated for the over-all management plan, while the other deals with the particular objectives of the monitoring system. It remains clear, however, that the objectives of the monitoring system design are a subset of the over-all management objectives.

Four considerations form the basis on which a rational design for a monitoring system can be achieved. They are the over-all management objectives, the particular monitoring system objectives as mentioned above, and the constraints imposed by the physical and financial conditions. Obviously, it would be impractical to formulate design objectives that are physically impossible to meet or that would require an unreasonably high proportion of the allocated project funds. The interrelation between these factors shown in Figure 1 indicates how these four aspects result in a monitoring system design strategy.

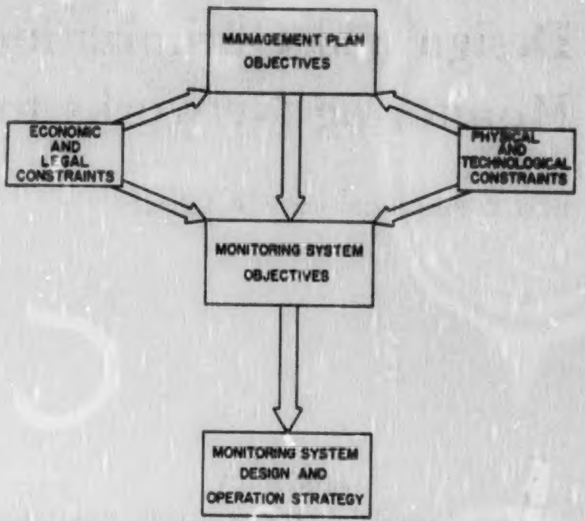
MANAGEMENT PLAN OBJECTIVES
ECONOMIC AND LEGAL CONSTRAINTS
PHYSICAL AND TECHNOLOGICAL CONSTRAINTS
MONITORING SYSTEM OBJECTIVES
MONITORING SYSTEM DESIGN AND OPERATION STRATEGY

Fig. 1. Evolution of monitoring system design.

With the establishment of a design strategy, an incremental stepwise design and implementation of successive phases of the information-gathering process is developed. The findings or outputs of each phase serve as input and design information for the following one. All steps are interrelated by feedback loops, that permit the maximization of the information base by multiple and inter-unit correlation and corroboration methods. For this general discussion five different phases of the monitoring system design and operation are identified (Figure 2).

DESIGN STRATEGY FOR MONITORING NETWORK

1. Preliminary Network Design and Information-Gathering Phase

During the formulation of objectives and the development of an operational strategy, it is necessary to review the physical and chemical conditions and the dynamics of the system in terms of the subsurface movement of any possible pollutants. The pathways and mechanisms of transport and dispersal that should enter into the design considerations are the following:

a. Infiltration from a pollution source to the subsurface.

b. Percolation through the unsaturated zone.

c. Transport and dispersion of disposed substances within the ground-water flow system.

In order to study the fate of pollutants in the

Table 1. Sources of Information

Maps (Areal Information)	*Records (Point Sources)*
1. Topographic	1. Well Logs
2. Aerial Photographs	2. Soil Borings
3. Geologic	3. Climatologic
4. Soils	4. Stream Data
Reports (Comprehensive Data)	*Miscellaneous*
1. Government Publications	1. Personal Communications
2. Journals	2. Current Research
3. Theses	a. Universities
4. University Research	b. Interest Groups
5. Environmental Reports	c. Government Agencies
	3. Field Surveys

subsurface, the relative importance of each one of these zones has to be properly assessed from a fluid motion point of view, and secondly from the perspective of possible importance of their environmental impact. This basic information is necessary to serve as framework to define and clarify the objectives, and to formulate feasibility and strategy considerations for the design and operation of the monitor system.

After the mechanisms have been considered and identified in a general way, the hydrogeologic setting has to be evaluated. Much of the general geology, meteorology and hydrology and a certain amount of site specific studies can be carried out at low-intensity, low-cost levels on the basis of pre-existing information. It is at this point that preliminary objectives, choice of methods and tools, and planning of the subsequent steps is carried out. An overview of sources of information and tools for independent information generation are listed in Tables 1 and 2.

2. Initial Installation and Testing Phase

Here the minimum number of observation and test wells are installed to give information about the general ground-water flow field. The information base is increased by intercorrelation of project well hydrographs to determine the quality of observations and the degree of confinement of the ground-water system. Intercorrelation gives some further idea about the persistence of the flow field configuration with time, and therefore serves as a basis for critical area designation. The bulk response of the system to precipitation and infiltration events can be estimated by correlation of well hydrographs with general climatologic data. Finally, correlation of project wells will give some idea about the lateral extent of the flow field and the possible vertical interconnection with deeper aquifer systems. For example, preliminary analyses can be carried out from sampling points that lie downstream along the principal axis of flow and dispersion. At this point close, intermediate and distant observation wells are adequate to assess the approximate extent of any existing contamination.

From drillers' logs and drill samples, a number of hydrogeological parameters can be estimated or determined in relatively inexpensive laboratory tests. These deal with flow properties such as permeability, relative permeability of immiscible phases, and retention capacity as well as adsorption and ion exchange properties of the soil material for the downward migration.

3. Completion and Verification Phase

In this phase, the observation well network is completed on the basis of information obtained

Table 2. System Evaluation Tools and Methods

	Geologic System	*Hydrologic System*
Direct:	1. Drilling 2. Grab Samples 3. Lab Analysis	1. Climatology Monitoring 2. Streamflow Monitoring
Indirect:	1. Geophysical (Shallow)	1. Water Budget Modeling 2. Watershed Modeling
	Hydrogeologic System	*Chemical System*
Direct:	1. Observation Wells/Piezometers 2. Pumping Wells 3. Suction Lysimeters 4. Laboratory Analysis 5. Aquifer Testing	1. Water Samples 2. Analysis of Samples 3. Ion Exchange and Adsorptive Capacity of Media
Indirect:	1. Flow Net Analysis 2. Analog and Digital Models	1. Isoconcentration Map 2. Geophysical (Density)

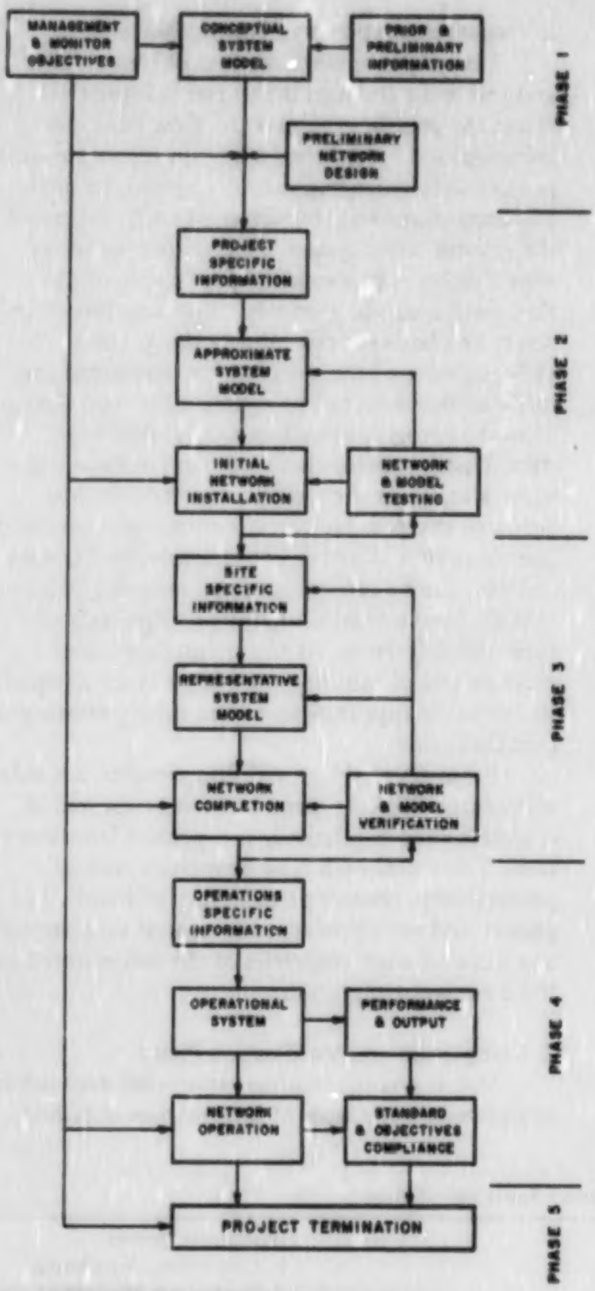
MANAGEMENT & MONITOR OBJECTIVES
CONCEPTUAL SYSTEM MODEL
PRIOR & PRELIMINARY INFORMATION
PHASE 1
PRELIMINARY NETWORK DESIGN
PROJECT SPECIFIC INFORMATION
APPROXIMATE SYSTEM MODEL
PHASE 2
INITIAL NETWORK INSTALLATION
NETWORK & MODEL TESTING
SITE SPECIFIC INFORMATION
REPRESENTATIVE SYSTEM MODEL
PHASE 3
NETWORK COMPLETION
NETWORK & MODEL VERIFICATION
OPERATIONS SPECIFIC INFORMATION
OPERATIONAL SYSTEM
PERFORMANCE & OUTPUT
PHASE 4
NETWORK OPERATION
STANDARD & OBJECTIVES COMPLIANCE
PROJECT TERMINATION
PHASE 5

Fig. 2. Flow diagram of network design process.

during the first two phases in order to meet project specifications. At the same time intercorrelation studies of Phase 2 are continued so that the existing data base can be refined.

The network density is increased (i.e. more wells) to give an adequate resolution to solve the problem at hand, such as more precisely defining the spatial extent of a pollution enclave.

A crude model of the system as formulated on the basis of information obtained in Phase 2 is now verified by checking whether the additional information obtained in Phase 3 corroborates the model. Furthermore, the quality and accuracy of the collected data is verified by the continued correlation of the monitoring network. On the basis of this information, sampling frequencies used in the next two phases can be established.

The principal consideration in the first three phases is to complete each step (or substep) with minimum expenditures in order to minimize redundant information.

4. Operational Phase

During the first three phases, the greatest frequency and number of tests, sampling operations and readings occur. These data are then used to establish the final operation program designed to give maximum efficiency by selecting only the most critical points at the lowest frequencies compatible with project objectives. During this operational phase, contingency funds must be available to increase or modify the monitoring network in case long-term observations change the initial short-term findings. It is during the operational Phase 4 that the importance of proper objective formulation, adequate system and process identification and most efficient monitoring network design and operation become obvious. The total number of observations and analyses are determined and thereby the cost of operations.

5. Project Termination Phase

An integral project phase that is often overlooked in the planning stage is the termination of the project. It is necessary to determine when sufficient information has been acquired. Too often neglect of this phase tends to prolong data collection beyond its managerial or engineering usefulness.

It is therefore necessary to consider this phase when objectives are formulated. At this early point the criteria should be established to determine when further monitoring is unnecessary.

APPLICATION OF PRINCIPLES

The basis for the preceding monitoring network development strategy results from over five years of monitoring and data acquisition of the surficial and ground-water conditions surrounding the University of Minnesota's chemical and special wastes disposal facility. For well over ten years, chemical and biological laboratory wastes were collected and transported to a disposal facility on relatively isolated University property, 25 miles south of Minneapolis and St.

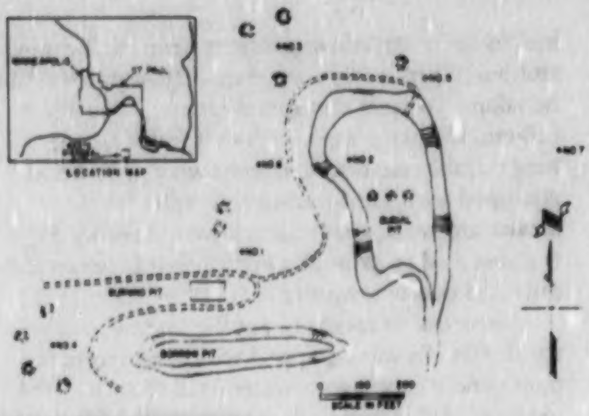

Fig. 3. Waste disposal site.

Paul (Figure 3). With the advent of federal and state regulations concerning the environment, the University found it necessary to reassess its disposal policies and evaluate the potential impact of the facility on the environment. Of particular concern was whether any of the waste had reached the ground water and if so, what was the extent of degradation.

The study that resulted was designed to address these problems. The early information-gathering phases, the incremental drilling and the maximization of data led to the methodology used for the design of a ground-water monitoring network. The advantages and shortcomings of the data collection and testing procedures used during the course of the study helped determine the inadequacies of the original methodology and led to the refined version discussed here.

THE WASTE DISPOSAL SITE

The disposal process for the chemical and special wastes involved collecting the wastes from the University laboratories once a week. The flammable and nonexplosive wastes were carefully removed from the truck and dumped into a 15-foot (4.5-m) deep disposal pit. A flare was then thrown into the pit from behind a protective barrier to ignite the flammable materials. Much of the flammable chemicals were volatilized by the process; however, residual material and nonflammable chemicals remained on the pit bottom ready for percolation of leachates into the subsurface.

PROJECT OBJECTIVES AND PRELIMINARY NETWORK DESIGN

On the basis of the disposal method and information on the type of wastes, certain project objectives were outlined. First, and of primary concern, was whether the residual wastes from the disposal areas had percolated through the 50-foot (15-m) thick unsaturated zone to the underlying drift aquifer. Second, if pollutants had entered the ground-water system, then determine the extent of the pollution plume and where it was going. Third, determine the rate of entrance of the pollutants into and their dispersal within the ground-water system. Finally, evaluate the interconnectivity between the drift aquifer and the more regional bedrock aquifer and establish if a potential regional hazard existed.

Given these objectives, the diverse nature of the wastes and the disposal practices, design of a ground-water monitoring network had to be flexible and capable of serving a number of functions. Specifically, the network had to be properly aligned to intercept a pollution plume and allow water-quality sampling. It also had to be structured to allow hydrogeologic analysis of the underlying flow system. To properly design the network, background information about the geology, and climatic conditions had to be collected. A list of tests and equipment needed to meet the project's objectives had to be formulated. In essence, the preliminary network design and information-gathering phase had to be completed.

GEOLOGIC SETTING AND PRIMARY HYDROLOGIC UNITS

The waste disposal site is located near the edge of the St. Croix moraine on the Rosemount outwash plain formed during the retreat of the Superior Lobe of the Wisconsin glacier (Ruhe and Gould, 1954). Having an areal extent of over 100 square miles, (260 m^2), this outwash plain serves as a drift aquifer of major proportions. In the vicinity of the waste disposal site, the outwash plain varies in thickness from 30 to 80 feet (9 to 24 m) and is composed of interbedded layers of fine to coarse sands and gravels. The surface is flat to gently rolling and pockmarked with depressions (possibly kettle holes). No defined drainage courses exist on the outwash plain. A clay layer of variable composition underlies the outwash plain and rests on the bedrock surface within the area surrounding the waste disposal site (Labno, 1973).

The bedrock surface has been dissected by preglacial streams and then filled with glacial debris forming an intricate network of buried valleys throughout the Minneapolis-St. Paul region. One of the larger buried valleys, cut nearly 400 feet (122

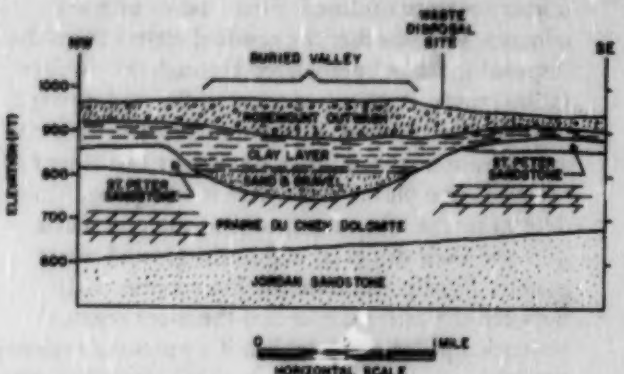

Fig. 4. Geologic setting.

m) into the bedrock, lies to the northeast of the study area and apparently controlled the preglacial drainage. As a result a series of smaller buried valleys are incised into the bedrock as tributaries of the larger valley. The waste disposal site lies on the flanks of one of these valleys (Figure 4).

The Prairie du Chien dolomite underlies the waste disposal site and is found at the base of the local buried valley. St. Peter sandstone overlies the Prairie du Chien dolomite and is found along the upper portions of the buried valley walls.

The Prairie du Chien dolomite and underlying Jordan sandstone are hydrologically connected and form a major bedrock aquifer in the region. The regional dip of the bedrock formations is towards Minneapolis and St. Paul forming a structural basin. This places the waste disposal site at the higher elevations of Prairie du Chien-Jordan aquifer and therefore near one of the aquifer's recharge areas (Norvitch *et al.*, 1973). The interconnection between the surficial Rosemount Outwash aquifer and the underlying Prairie du Chien-Jordan aquifer becomes important since the contamination of the drift aquifer could affect the more regional bedrock aquifer.

DEVELOPMENT OF THE MONITORING NETWORK

With the knowledge gained from preliminary investigations of the regional and local geology, a general idea of the ground-water flow direction in both the drift and the Prairie du Chien-Jordan aquifer could be estimated. Three observation wells were installed and water levels confirming the flow directions were taken. Plans were then delineated for the installation of four other wells.

Each of the wells was designed and located to satisfy certain project objectives. Since the determination of the extent if any of a pollution enclave was of primary importance, certain wells had to be located downgradient from the burning and burial pits so that water-quality samples could be taken. To meet this specification, 7 four-inch (10-cm) diameter wells with two-foot (.61-m) long suitably sized well screens were drilled and equipped with hand pumps and water-level measuring ports. One bedrock well (number 5) was installed to provide a correlation between the drift and bedrock aquifer.

In order to facilitate aquifer testing, one of the drift wells was equipped with a submersible pump and a continuous water-level recorder. The location of this well with respect to the others had to be carefully planned so as to obtain the best results during an aquifer test. In addition, a U.S. Weather Service Standard recording rain gauge was installed adjacent to the well capable of measuring rainfall intensity and duration. This data plus the continuous barometric and other meteorological data recorded at the University's Agricultural Station, located one mile west, were of importance to the total ground-water resource evaluation.

Once the layout of the wells and the type of equipment was established, the remaining four wells were installed. Each well was developed with the hand pump, and water levels were noted. After a few weeks, a monitoring program was determined. Periodic review of the accumulated data helped determine the adequacy of the program and the development of any problems.

During the completion and verification stage, biweekly water-level measurements were taken and plotted. Seasonal water-level fluctuations were noted and all wells, with the exception of two, fluctuated in a similar manner suggesting a homogeneity of the flow system. To more closely monitor the changes of the drift aquifer, three continuous well records were used to note instantaneous changes. Shortly after their installation, it became readily apparent that the drift aquifer was homogeneous. The instantaneous hydrographs (Figure 5) matched very closely, signifying an intercorrelation of project wells and a unified response to changes in the ground-water system.

In contrast, wells 3 and 5 fluctuated erratically

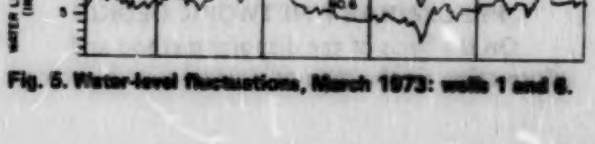

Fig. 5. Water-level fluctuations, March 1973: wells 1 and 6.

on a daily and weekly basis, but for the most part followed the established seasonal variations of the other wells. Examination of the instantaneous hydrographs demonstrated that erratic fluctuations were common and that they could be attributed to barometric fluctuations. The silty sandy matrix surrounding well number 3 apparently was sufficient to confine the well, thereby increasing its barometric efficiency to a point of being quite similar to the bedrock well number 5. In the latter well, the 20 feet (6 m) of overlying clay material was of sufficient size and thickness throughout the study area to confine the Prairie du Chien-Jordan aquifer to create a very noticeable barometric response.

The importance of this intercorrelation of wells becomes apparent when a flow net analysis of the system is conducted. The drift aquifer is considered homogeneous, therefore implying a constant flow direction. However, the inclusion of well number 3 water levels causes a change in the flow net construction and ultimately the flow direction. This would then suggest a greater potential spread of pollutants than actually existed.

In addition to the intercorrelation of wells, tests measuring the aquifer response to meteorologic events were conducted. Such factors as aquifer response to precipitation, snow melt and barometric changes could be noted with the continuous well recorders. Analysis of the instantaneous well hydrographs taken throughout the year suggested that a direct response to a rainfall event could be noted during periods of recharge while no response to rainfall was noted during ground-water recession periods. This, when combined with information gained on aquifer recharge, indicated when infiltration was the greatest and possibly when the greatest amount of leachate might reach the ground water.

Throughout the testing phase, water samples were taken monthly and checked for evidence of a pollution enclave. The results from these preliminary samples proved to be encouraging. No signs of a pollution enclave were noted. Water samples taken during an aquifer test using well number 6 similarly showed no significant change in water quality. The water-quality monitoring program was then geared to these results.

As was previously discussed, long-term monitoring will sometimes alter initial findings. The lack of evidence for a pollution enclave raised the question as to what depth had leaching occurred. The 50 feet (15 m) of unsaturated sand and gravel apparently was a sufficient buffer for attenuation of leachate to the ground water, yet due to the nature of the porous media, some movement of pollutants should occur.

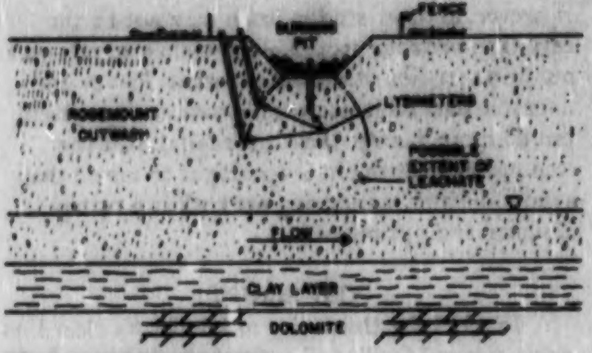

Fig. 6. Lysimeter placement.

To evaluate the situation, soil samples were taken from various depths below the burning pit and suction lysimeters installed in the holes. Chemical analysis of the soil and water samples demonstrated that the wastes had leached from the burning pit and decreased in concentration with depth. Additional lysimeters placed outside of the pit (Figure 6) at an angle indicated that the pollutants were fanning out in a bell-shaped manner. Field and laboratory studies by Saint (1973) discussed the potential factors affecting leachate movement and dispersion below the burning pit and ultimately concluded that no serious threat to the ground-water system had been detected.

Monitoring of the quality and the hydrogeologic conditions of the aquifer continued for a number of years and eventually was terminated. No adverse impacts on the ground-water system were noted.

THE PROJECT IN RETROSPECT

Application of the five phases developed at the beginning of this paper to the waste disposal study is easy in retrospect as are those portions of the study that could have been better planned. For example, the pumping test and preliminary flow net analysis were conducted before the intercorrelation of wells had been completed. Although the data was not unusable, it did result for a while in some erroneous conclusions. In another case, it was nearly two years after the project began that the well number 1 measuring point elevation was found to be off by nearly one foot. This gave anomalous flow direction 70° off the actual flow.

The above are problems that were rectified and had no effect on the conclusions of the study.

However, in other studies, such may not be the case. The need for proper planning in ground-water pollution studies should be apparent. Faulty and unusable data collected from a monitoring system can do more damage than lost capital. It is the responsibility of the hydrogeologist to insure that a ground-water monitoring network is properly designed and optimized.

POST SCRIPT

In 1975 the University of Minnesota closed its waste disposal facility after finding other practical and safer means of disposal. The residual wastes in the burning and burial pits will remain and continue to leach into the environment. However, based on the research conducted by the various University departments, a serious hazard is not expected. Should the wastes eventually leach to the ground water, the dispersive characteristics of the aquifer are sufficient to eliminate any major problems. The remote nature of the site is another plus factor. The nearest well downgradient is over a mile and a half away. Finally, the clay layer overlying the Prairie du Chien dolomite is known to be quite extensive locally and can prevent direct connection with the bedrock aquifer.

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(Mr. Labno's Discussion Questions were lost.)

Tracing Sewage Effluent Recharge — Tucson, Arizona[a]

by Thomas R. Schultz[b], Jeffery H. Randall[b],
L. G. Wilson[c] and Stanley N. Davis[b]

ABSTRACT

Dry washes or river beds are often used by southwestern communities to dispose of treated sewage effluent. Because many of these communities rely on ground water as a water supply, there is concern that this disposal practice may contaminate local aquifers. This has led to implementation of monitoring and tracing programs to quantify effluent and ground-water interactions and to development of efficient, easily used predictive models.

The treated sewage effluent from the City of Tucson treatment plant has historically been used for irrigation and/or discharged to the normally dry Santa Cruz River. Numerous sampling programs have been undertaken to quantify the chemical quality, temperature, and microbiological activity of the ground water in the area near the Santa Cruz. Ground-water regions with high chloride and nitrate concentrations tend to be associated with areas irrigated with sewage effluent. Quality degradation due to channel recharge is not as evident because the effluent recharge is restricted by fine materials plugging the channel deposits. Recharging water tends to mound near the contact between the Recent and Fort Lowell formations spreading laterally more rapidly than downward.

A new tracer, trichlorofluoromethane (trade name Freon 11, Cl_3CF) with applications similar to environmental tritium is being evaluated. Cl_3CF enters the hydrologic cycle when it is partitioned between the gas and liquid phases during raindrop formation. Cl_3CF in water samples is separated and quantitatively measured by a gas chromatograph with pulsed electron-capture detector.

Preliminary Cl_3CF analyses of ground water along the Santa Cruz do not correlate with nitrate values because mixing and increasing atmospheric Cl_3CF concentrations were not accounted for. However, the presence of Cl_3CF in the ground water indicates recent recharge. Predictive modeling will be implemented using Cl_3CF and a finite-state mixing model.

INTRODUCTION

It is a common practice in southwestern communities to utilize dry washes or river channels for the disposition of wastewaters. The City of Tucson, Arizona has, for example, discharged secondary treated effluent to the normally dry Santa Cruz River since 1951. Because Tucson relies completely on ground water, withdrawing about five times the natural recharge rate (Arizona Water Commission, 1975), there is concern that this effluent disposal practice may lead to ground-water contamination. Consequently, local water managers are currently (1) examining monitoring and tracer techniques to quantify the interactions of effluent and ground water, and (2) evaluating the role of efficient and easily-implemented models to predict such interactions.

The area discussed in this paper is located in the northwest corner of the Tucson Basin, covering about 60 square miles (155 square kilometers) and includes 25 river miles (40 kilometers) of the Santa Cruz River (Figure 1). At present, this particular reach of the River receives the entire discharge of the City of Tucson treatment plant, amounting to 31,000 acre-feet (38 million cubic meters) in 1974 (Dye, 1974). The main downstream ground-water uses are for irrigation and individual water supplies,

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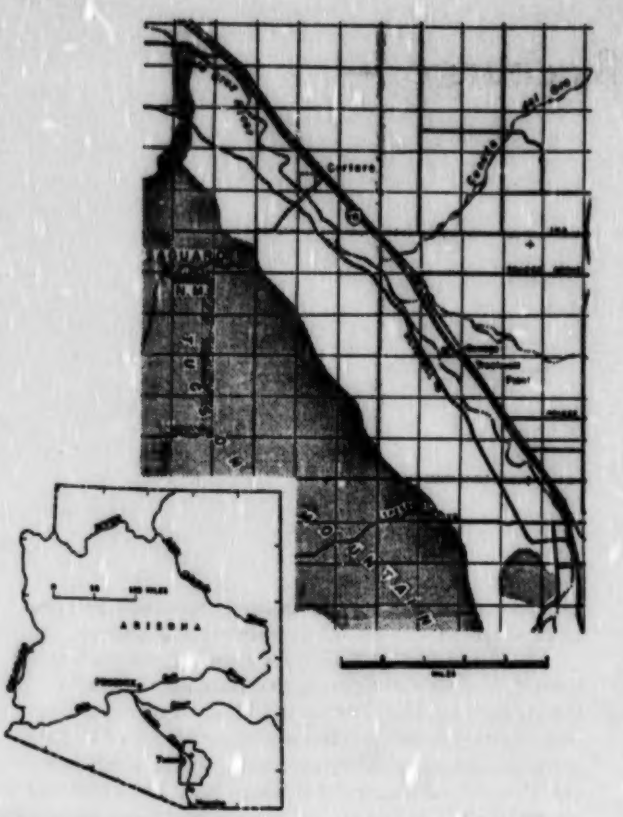

Fig. 1. Santa Cruz River study area, northwest corner of Tucson Basin (inset).

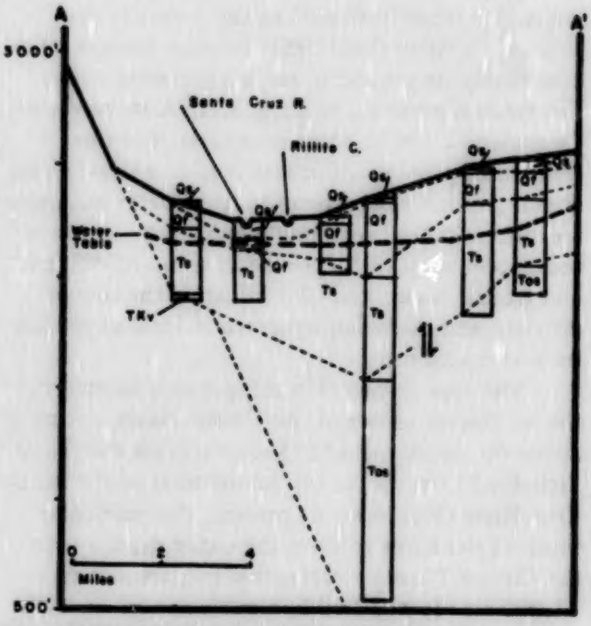

Fig. 2. Diagrammatic cross section of the Santa Cruz River study area (adapted from Davidson, 1973).

although with future growth of Tucson it is expected that the major usage will be for municipal water supply.

The purpose of this paper is to illustrate some of the traditional monitoring-tracer programs used in the study area and to describe a potential new tracer, trichlorofluoromethane (trade name Freon 11), for tracing effluent movement and for use in conjunction with predictive models.

HYDROGEOLOGY

The Tucson Basin lies within the Basin and Range Province of the southwestern United States where downfaulted or rifted blocks, bounded by steeply-dipping normal faults, form the valleys. The material filling the intermontane basin consists of locally derived clastic material or alluvium deposited since the Oligocene (about 40 million years ago). A secondary process of erosion has formed three terrace levels in the Tucson Basin (Davidson, 1973). These are from oldest to youngest the University, Cemetery, and the Jaynes terraces.

Davidson (1973) divided the sediments comprising the aquifer in the study area into three major units as shown in Figure 2. The Tinaja Formation, Ts, consists of 200 to 500 feet (60 to 150 meters) of sandy gravel with 10 to 50 percent silt which unconformably overlies the Pantano, Tos. The Fort Lowell Formation, Qf, unconformably overlying the Tinaja, is the most productive aquifer in the Basin (Davidson, 1973). In the study area this formation consists of 50 to 150 feet (15 to 45 meters) of loosely packed sands and gravels, the thinnest zones underlying the Santa Cruz River channel. The most common lithology is granite, gneiss, and locally derived volcanic clasts in a sand and montmorillonite matrix. Overlying the Fort Lowell along the Santa Cruz River are the Recent stream-channel deposits, Qs, comprised of clean, reworked sand and sandy gravels which grade laterally to a thin veneer of sheet flow and alluvial fan sediments. The thickness of the Recent deposits below the River channel is 40 to 100 feet (12 to 30 meters), averaging about 80 feet (24 meters).

Because most wells in the area are screened throughout the entire saturated thickness and penetrate into the Tinaja, reliable values of individual formation transmissivities do not exist. However, Davidson (1973) and others estimated the transmissivities in the study area to be the following: 2500 to 7000 $ft^2\ day^{-1}$ (230 to 650 $m^2\ day^{-1}$) for the Tinaja and Fort Lowell, and 10,000 to 40,000 ft² day⁻¹ (930 to 3800 m² day⁻¹) for the Recent channel-alluvium.

The Recent deposits, though very permeable, are dry in most of the Basin and have only about 10 to 20 feet (3 to 6 meters) of saturated thickness below the Santa Cruz. Their major hydrologic significance is as a recharge conduit for flood and sewage-effluent flows and excess irrigation waters.

The local ground-water flow direction is to the northwest with a gradient of about 20 feet per mile (3.8 meters per kilometer) as shown in Figure 3. The ground-water outflow of the study area is constricted by the surface and subsurface expression of the northern end of the Tucson Mountains. The low permeability of the volcanic rocks forming the constriction tends to force the ground water closer to the land surface.

TRADITIONAL MONITORING AND TRACING TECHNIQUES

Historically, effluent from the City of Tucson treatment plant has been utilized for irrigation and/or discharged to the Santa Cruz River. However, since 1971 the entire flow has been discharged to the River. Concern over the possibility of ground-water contamination, as a by-product of these disposal practices, generated several monitoring programs. Davis and Stafford (1966) reported the results of a sampling program in 1965-1966 characterizing the chemical quality, temperature, microbiological and virological levels in 105 wells. Organics were evaluated via the carbon chloroform extract (CCE) method. Three regions downstream of the plant manifested the presence of ABS in well waters—a definite indication of effluent recharge. Nitrate levels were also greater than 45 mg/l in a large number of samples. Only 10 wells showed the presence of coliform or fecal streptococci.

A study by Cluff *et al.* (1972) involved sampling irrigation wells in the Spring of 1966 and 1970. In 1966, 23 wells were found with nitrate levels between 4 and 122 mg/l. Individual concentrations in the same wells in 1970 ranged between 5 and 51 mg/l. Matlock *et al.* (1972) reported on a sampling program using 400 wells in the area. Their nitrate concentration maps were similar to those presented by Davis and Stafford (1966), with high values (up to 45 mg/l) in the region irrigated by effluent. As with the distribution of ABS observed by Davis and Stafford (1966), the levels of boron from LAS type detergents were found to be in excess of background in the vicinity of irrigated areas and the Santa Cruz River.

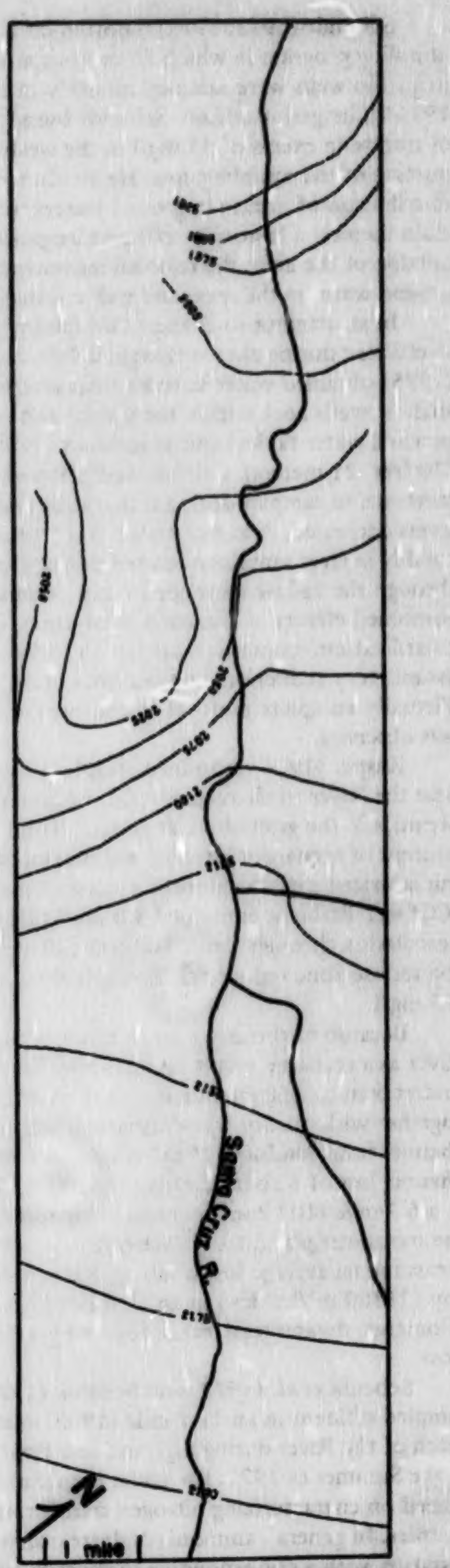

Fig. 3. Computer plotted potentiometric map of the Santa Cruz River study area in feet above mean sea level.

Schmidt (1972, 1973) reported on a sampling program in which 20 large-capacity irrigation wells were sampled monthly in the 1971-72 irrigation season. Schmidt found levels of nitrate in excess of 55 mg/l in the western portion of the sampling area. He attributed the distribution of nitrate in ground waters to three main factors: a history of effluent irrigation in a portion of the area; the regional movement of ground water in the area; and well construction.

In an attempt to characterize quality changes in effluent during river recharge, Wilson *et al.* (1975) obtained water samples from a number of shallow wells both within the vadose zone (i.e., in perched water tables) and at the water table (70 feet, 21 meters). Calcium and magnesium levels increased in samples during recharge but sulfate levels decreased. A loss of about 90 percent of the total-N in river samples occurred during flow through the vadose zone apparently because of the combined effects of filtration of organics, ammonia volatilization, ammonia sorption on clays, assimilatory reduction, and denitrification. Virtually complete removal of coliform organisms was observed.

Kasper (1976) examined samples of well water near the River to characterize the movement of organics in the ground-water system. Total organic content of recharged effluent was measured by the miniaturized carbon chloroform extract method (CCE-m). Effluent contained 3.8 mg/l CCE-m, but percolation through about 100 feet (30 meters) of the vadose zone reduced CCE-m values to less than 0.5 mg/l.

Because of the importance of the Santa Cruz River as a recharge source, studies have been undertaken to characterize intake rates of effluent together with quality transformations during channel flow. Matlock (1966) found an average channel loss of 6 ac-ft/acre/day (18300 m^3/ha/day) in a 6.3-mile (10-kilometer) reach downstream of the treatment plant. Later, Sebenik *et al.* (1972) measured an average loss of about 7.0 ac-ft/acre/day (21300 m^3/ha/day) in an 11-mile (17.5-kilometer) downstream reach following a flood flow.

Sebenik *et al.* (1972) and Sebenik (1975a) sampled effluent in an 11.7-mile (19-kilometer) reach of the River during high and low flow periods in the Summer of 1971. Particular emphasis was placed on characterizing nitrogen transformations in samples. In general, ammonia-N decreased with distance, with a corresponding increase in nitrate-N during both low and high flows. Sebenik (1975b) examined the relationships of dissolved oxygen (D.O.) and biological oxygen demand (B.O.D.) in effluent within the channel during high and low flows of effluent. He found that D.O. levels were higher at low effluent flows and conversely B.O.D. values were higher during high flows. He related these results to waste loadings, flow conditions, phytoplankton growth and nitrification.

Water quality data (Courtesy Department of Soils, Water and Engineering, University of Arizona, Tucson, and Cortaro Water Users Association), collected during 1971 from 280 wells near the Santa Cruz River, were mapped by the authors to indicate the spatial movement of the recharging sewage effluent. The water quality parameters chosen were calcium, magnesium, sodium, chloride, bicarbonate, nitrate, electrical conductivity, and total dissolved solids. The data were plotted and contoured using an advanced computer graphics package at the University of Arizona. The resulting maps compared favorably with those of Matlock *et al.* (1972). The aerial distribution of chloride and nitrate, chosen to be representative, are shown in Figures 4 and 5.

Shallow water quality patterns in the area strongly reflect local hydrogeological controls. In particular, the channel deposits have a hydraulic conductivity about one order of magnitude greater than the underlying Fort Lowell and Tinaja formations. This causes recharging water to mound near the interface with the Fort Lowell and to spread laterally more rapidly than downward. The spreading is reflected in the ground-water quality of the selected wells (Table 1) plotted in the Piper diagram (Figure 6). The shallow wells, which only penetrate the upper zones of the Fort Lowell (SF1A1, RRCY1, and IRA21), have a water type similar to the effluent water in the Santa Cruz (SCRR1). Deeper wells deriving most of their water from the Fort Lowell and Tinaja have a similar water type, but different from the shallow wells, manifesting long-term vertical mixing of effluent and indigenous ground water.

The areas of high chloride and nitrate concentrations, shown in Figures 4 and 5, tend to correspond to areas that have been irrigated with sewage effluent. Availability of an inexpensive, abundant supply led to overirrigation with two to three feet of effluent per year in excess of consumptive use requirements (Schmidt, 1973). Because the channel deposits underlying the agricultural areas are highly permeable, over-irrigation caused a large surface area of the aquifer to receive effluent recharge. The problem was

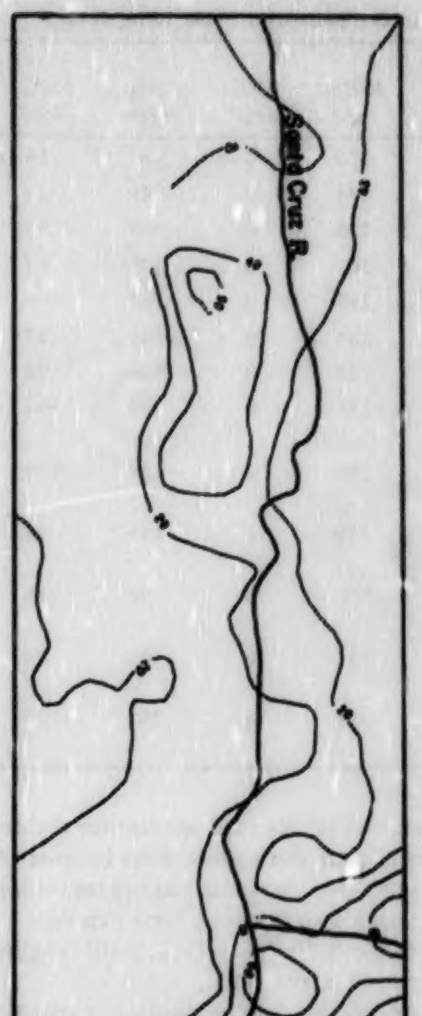

Fig. 4. Computer plotted isograms of nitrate in mg/l for the Santa Cruz River study area.

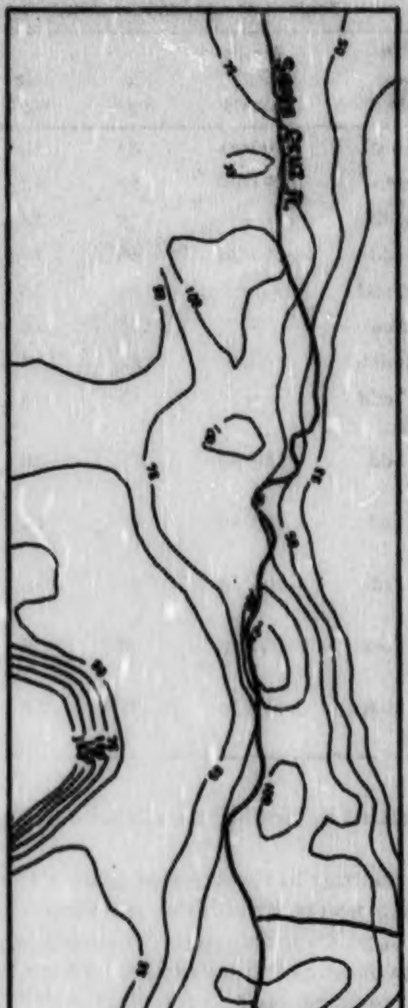

Fig. 5. Computer plotted isograms of chloride in mg/l for the Santa Cruz River study area.

Table 1. Representative Water Quality Analyses of Wells in the Santa Cruz River Study Area

Location (Tn. Rn.) Section	*Depth feet/ meters*	*Ca mg/l*	*Mg mg/l*	*Na mg/l*	*K mg/l*	*HCO_3 mg/l*	*CO_3 mg/l*	*SO_4 mg/l*	*Cl mg/l*	*NO_3 mg/l*
(13-12)2add	300/91	83	16	108	–	270	0	185	54	18
(13-13)5bba	294/90	57	12	24	–	144	0	49	24	4
(13-13)6cdd	–	119	22	92	–	293	0	298	91	–
(13-13)7cdd	250/76	60	17	163	13	307	0	130	84	40
(13-13)27cdd	–	124	30	127	13	305	0	257	109	23
(13-13)28cd	–	115	16	183	–	264	0	310	145	–
(13-13)34dbb	–	82	14	144	–	225	0	240	98	5
(13-13)17cdd (SCRR1)	0	51	8	142	0	266	0	149	104	3
(13-12)1bdd (IRPW1)	280/82	125	20	120	6	381	0	25	105	4
(13-12)1bdd (IRA21)	100/30	41	13	128	13	259	0	140	94	1
(13-13)17cda (RRCY1)	100/30	51	6	122	14	288	0	84	111	3
(13-13)17cca (SF1A1)	100/30	48	9	116	15	251	0	123	85	1
(13-13)28bdc (SF2A1)	100/30	125	20	186	–	371	0	252	190	2

compounded by cascading water within irrigation wells.

In contrast to the irrigated areas, effluent recharge from the River is not as evident from quality data. River seepage is controlled primarily by the hydraulic conductivity of the river bed. Several researchers (e.g., Sebenik *et al.*, 1972) have reported that intake rates are restricted through the channel bottom during low flows because of plugging by fine inorganic and organic sediments. Occasional flooding scours these deposits permitting substantial recharge until clogging again retards intake rates.

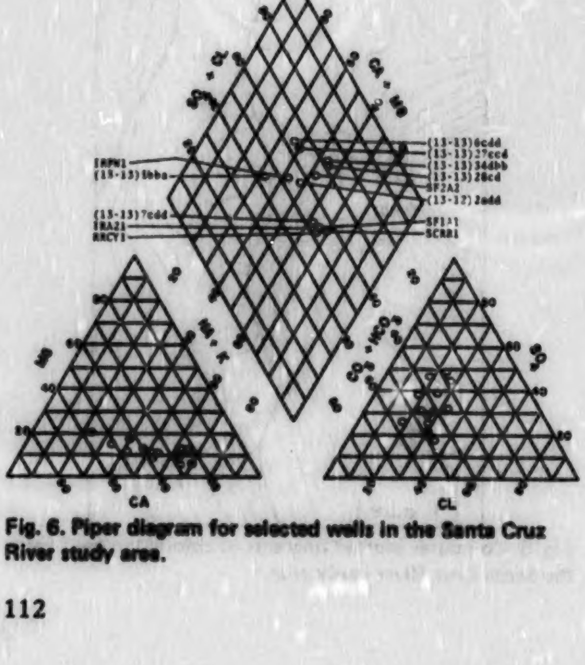

Fig. 6. Piper diagram for selected wells in the Santa Cruz River study area.

The chloride ion distribution, a traditional tracing method, is further complicated due to the high natural background in the study area. The two anomalous areas to the west of the River have been correlated to lacustrian mud and siltstone deposits which are frequently associated with evaporites in the Basin and Range Province.

FREON TRACING APPLICATIONS

A new tracer, trichlorofluoromethane (Cl_3CF), has been evaluated for use in hydrological studies (Thompson *et al.*, 1974). Applications of this tracer are similar to tracing applications of environmental tritium. Trichlorofluoromethane is a commercially produced compound manufactured since the 1930's, primarily by E. I. duPont de Nemours and Company under the trade name of Freon 11. Cl_3CF has been widely used as an aerosol propellant, working fluid in refrigeration systems, blowing agent in plastic foams, and as an industrial solvent. This compound and others like it have received much attention recently due to their postulated role in the stratospheric destruction of ozone.

Tropospheric introduction of Cl_3CF is a function of its intended usage. The residence time between manufacture and use of aerosol propellants is short. Rowland and Molina (1975) estimate that 90 percent of the propellants are released within 6 months. The residence time for blowing agents and solvents has been estimated to be near that of aerosol propellants. Tropospheric introduction of Cl_3CF is not uniform, being greatest around urban areas in industrial nations (Hester *et al.*, 1974). However, global tropospheric mixing is rapid (on the order of months) and uniform. The distribution of artificially produced radioactive krypton-85 as a Cl_3CF analog (Telegradas and Ferber, 1975) shows relatively uniform mixing within both hemispheres with only a 20 percent reduction in concentration in the southern hemisphere.

The introduction of Cl_3CF into the hydrologic cycle occurs when it is partitioned between the gas and liquid phases during precipitation. There are two possible mechanisms of incorporation in precipitation, rain-out and wash-out. Rain-out involves solution of the gas during drop formation and wash-out involves solution as the raindrop falls. Based on other trace gases (SO_X and NO_X) in precipitation, rain-out is predominant (Moyers, 1976). If rain-out is the predominant mechanism, then the Cl_3CF concentration in precipitation will be in equilibrium with the global atmospheric distribution rather than near surface anomalies in urban areas. The concentration of Cl_3CF in recharging water is therefore proportional to the tropospheric concentration at that location and time.

Cl_3CF in water samples is separated and quantitatively measured by a gas chromatograph with pulsed electron-capture detector. The chromatographic system developed at the University of Arizona is a modification of an Indiana University design (Thompson 1975). The system has several features distinguishing it from the analytical quantification of carbon-14 or tritium: it is fast and easily operated; allows sample concentration; and most importantly, is field operable.

The major demonstrated advantages of Cl_3CF as an environmental or artificially injected tracer can be summarized as follows: no quality degradation (a problem with large concentrations of chloride tracers); extremely low toxicity, thereby minimizing health hazards; low sorptivity on clays due to its low surface energy (a problem with organic dyes); quick, economical field-operable detection with equipment less expensive than for most other tracers; a detection limit of 10^{-13} grams per second; a steady known build-up in the atmosphere (atmospheric variability is a major problem with tritium); and no loss through decay as with radioactive isotopes.

To develop a predictive tool more quantitative than the contouring described earlier requires a computer model. Mass flow of water can now be handled by numerous computer modeling algorithms (e.g., Prickett and Lonnquist, 1971). Modeling chemical mass flow is more difficult and is still in its infancy. A finite-state mixing cell model (FSM) has recently been developed at the University of Arizona (Campana, 1975) which accounts for chemical and water mass-flows without requiring assumptions about dispersion coefficients. The FSM has been calibrated and verified for the Tucson Basin using carbon-14 as a tracer (Campana, 1976) and for the Edwards Limestone in Texas using tritium. A more detailed FSM model is being calibrated for the study area shown in Figure 1 using the collected historical chemical data. Once calibrated, Cl_3CF data will be used in the FSM to demonstrate the tracer's usefulness as an age dating tool in sewage effluent tracing (Randall and Schultz, 1976).

Cl_3CF concentrations, from the preliminary analyses shown in Table 2, were mapped and correlated with the corresponding nitrate values. The resultant map and correlation analysis indicate that there is no apparent correlation between the two. This is not surprising because mixing has been neglected and more importantly, the historically increasing Cl_3CF concentration in the recharging water has not been accounted for. However, because Cl_3CF is an anthropogenic (man-made) compound, its very presence in the ground water near the Santa Cruz River indicates that recharge has taken place since the 1930's. This kind of qualitative information can be useful in delineation of regional or local recharge areas where other methods have proved impractical. Furthermore, it is expected that Cl_3CF when utilized in a finite-state model will provide quantitative information suitable for predictive modeling.

ACKNOWLEDGMENTS

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Table 2. Trichlorofluoromethane (Cl_3CF) Concentrations in Southern Arizona

Location (Tn.Rn.) Section	*Sample Type*	*Concentration (picograms/ml)*
(12-12)16bbb	Well	0.28
(12-12)16bbb (duplicate)	Well	0.13
(12-12)17ddd	Well	6.0*
(12-12)21bab	Well	0.62
(12-12)21bdd	Well	12.*
(12-12)25cac	Well	0.19
(12-12)26dba	Well	0.13
(12-12)26dbd (First)	Well	5.6*
(12-12)26dbd (Second)	Well	14.*
(12-12)36bab (First)	Well	3.9*
(12-12)36bab (Second)	Well	0.36
(13-13)35adb (A-35)	Well	3.4*
(13-13)16ddd (Z-2)	Well	2.8*
(13-13)16bca (Z-5)	Well	18.*
(12-13)31ccc (Z-13)	Well	17.*
(12-13)31dcd (Z-14)	Well	2.1*
St. David, Arizona	Well	<0.02
Santa Cruz River @Silverbell Rd.	Surface Water	0.08
Tucson	Air(1976)	0.70

* Possible sample contamination.

Nemours and Company. We gratefully acknowledge the assistance of Glenn Thompson of the Department of Geology, Indiana University, in making the preliminary Cl_3CF analyses.

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DISCUSSION

The following questions were answered by Stanley N. Davis after delivering his talk entitled "Tracing Sewage Effluent Recharge – Tucson, Arizona."

Q. *If there is no ion, normally analyzed for, to pose a problem in the usability of the ground water, why would you want to stir up the public by using a tracer to prove that the water contains sewage?*
A. This question is a very good general question, related to social and political justifications of ground-water studies. As a purely scientific answer, if general knowledge can be gained by hydrochemical studies, this is justification enough. In the case of the Santa Cruz River north of Tucson, it is common public knowledge that effluent is entering the subsurface. Numerous studies have been published and are also available to the public. Our interest is in testing new techniques in an area which is already quite well known. We do not anticipate that our studies will add to public anxiety.

Q. *What are typical freon concentrations in domestic sewage effluents?*
A. Insufficient measurements have been made to date. No typical values can be given. The next-to-the-last analysis given in Table 2 is of effluent that has entered the Santa Cruz River. The value given is lower than would be expected if it were in equilibrium with the atmosphere. Another analysis made at the City of Tucson sewage plant in April of 1976 also showed that the effluent was undersaturated with respect to atmospheric Freon-11.

Q. *Do the Tucson area waste treatment plants chlorinate their effluent prior to discharge into the receiving stream? If so, could you also look for chlorinated organics present in the underlying ground-water body?*
A. The Pima County Ina Road plant which discharges approximately 10 mgpd does not chlorinate its effluent. Effluent from the City of Tucson facility receives some chlorine. Chlorination has been sporadic in the past. At present, no one is analyzing for chlorinated organics in ground water; however, it would be feasible using analytical procedures similar to ours that incorporate the electron-capture detection system.

Q. *Many papers state that "a few feet of aerated zone thickness beneath a disposal system will remove large quantities of waste constituents indefinitely." Does your study and/or experience support or refute this frequent conclusion?*
A. The present study does not contribute information to answer this question. However, past studies cited in our paper do give some information. Some waste constituents such as chloride would not be removed; others such as very small traces of certain heavy metals might be removed for periods of several hundred years. Of course, biological and suspended solids can be mechanically filtered by underlying sediments.

Q. *You mentioned that in arid environments, potassium tends to travel great distances. Why is this so?*
A. The possible use of potassium was mentioned in the oral presentation but is not discussed in the written paper. Potassium levels in the Tucson effluent most commonly range from 15 to 20 mg/l, or somewhat higher than the native ground water. The native ground water in arid regions, however, is also high in potassium, commonly from 5 to 15 mg/l. This suggests, in turn, that many adsorption sites are already occupied by potassium ions and that the distribution coefficient would be relatively low with the transport correspondingly more efficient.

Q. *How much time and what is the expense to run a single fluorocarbon analysis?*
A. Once the sample has been collected in the proper container, the process of stripping the Freon-11 and passing it through the gas chromatograph takes less than 10 minutes. On-site duplicate samples should be run to check for consistency of data. A typical field visit to a site, including analyses, could take from 20 minutes to an hour provided the water is readily sampled and all components of the instrument are functioning. This is based on the experience of Glenn Thompson of the Department of Geology, Indiana University, who is presently making routine analysis of surface-water samples. He can be contacted for further details.

The cost of field analyses has not been estimated; however, it should be limited to operator time and transportation. There are very minimal expendable costs. Based on our experience at the University of Arizona, the cost of the electronics package and the chromatographic and detector system, including parts and labor, is on the order of $2500. A strip chart recorder is also convenient to supply a backup record and visual aid.

Monitoring Cyclic Fluctuations in Ground-Water Quality[a]

by Wayne A. Pettyjohn[b]

ABSTRACT

The chemical quality of water in many shallow and surficial aquifers exhibits cyclic fluctuations. These fluctuations are caused by the intermittent flushing of contaminants into the ground during recharge events. The contaminants may be natural or reflect man's activities, particularly waste disposal schemes.

Over the past 12 years an oil-field brine contaminated aquifer in central Ohio has been monitored. Data from three closely-spaced wells tapping selected parts of the aquifer indicate that brine is flushed into the ground during recharge events and that each contaminated mass maintains much of its integrity as it sinks to the bottom of the aquifer and then migrates laterally to the adjacent river. The most concentrated mass that covers the largest area infiltrates during the spring recharge period, but less concentrated and smaller masses may occur any time rainfall is sufficient to overcome the soil-moisture deficiency.

Because of the cyclic nature of recontamination events, care and common sense must be exercised in the extrapolation of quality data, particularly in regard to estimation of contaminant flushing rates.

INTRODUCTION

Within the past 10 years water pollution investigations, reports, and legal actions have far exceeded those of previous decades. In the literature, ground-water pollution case histories have become commonplace where before 1965 they were rare. Despite our having become much more aware of ground-water pollution, however, a great deal still remains to be learned and documented. A particular need is the development of adequate ground-water quality monitoring techniques that can provide us with information on the temporal and areal migration of contaminants so that we can develop accurate predictive models.

Samples of well water obtained for quality analyses may be collected only once, now and then, or perhaps on a regular schedule. The literature indicates that even during the monitoring of contaminated aquifers, samples may be collected only during a set time period or until the concentration of the contaminant begins to decrease. Furthermore, such important data as depth to water, well depth, and construction details commonly are not recorded.

We may not realize the pollution potential or that which is occurring in an area merely on the basis of a one-time collection period. We must consider the contaminant both in time and space.

The lack of many of the basic details mentioned above does not permit an investigator to accurately monitor a contaminated aquifer, determine when it may be flushed, or establish legal responsibilities. In most instances, we complain of a lack of information and stringent time constraints on data collection, monitoring, and perhaps well construction. Only in rare circumstances will these problems be overcome. Generally, we better perceive what should have been done only long after the project has been completed. Certainly there is no substitute for sound planning.

The purpose of this Symposium is to examine various methods that have been used to clean up ground-water pollution. Determination of the effectiveness of these techniques, to a large extent, depends on adequate monitoring, which must be accomplished in such a manner that the distribution of contaminants can be detected and predicted in both space and time.

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It is generally believed that the quality of ground water changes very slowly, if at all. This may be true in the majority of cases, at least when we deal with deep aquifers, those not contaminated, or those not greatly stressed by pumping or construction. On the other hand, many surficial or shallow aquifers are prone to rapid changes in quality, the effect being measureable within hours or days. Because rapid and perhaps repeated changes in quality can occur, we must be prepared to predict these events and understand the principles involved so that we can deal with them from legal, technical and practical viewpoints.

The chemical quality of water in many shallow aquifers exhibits cyclic fluctuations that have seldom been noticed, rarely reported, and apparently never described in detail. The purpose of this report is to shed some light on this phenomenon and to describe the vertical, horizontal, and temporal migration of contaminants introduced into a shallow aquifer by certain waste-disposal methods, agricultural techniques, and natural phenomena so that we can develop adequate ground-water quality monitoring techniques. A few examples to illustrate how some traditional methods can lead to erroneous conclusions will also be presented.

The major cause of cyclic quality fluctuations is intermittent infiltration of water-soluble substances that lie mainly on the land surface or in the ground above the water table. These substances include waste materials, such as brine and chemicals, nitrogenous materials of natural or man-made origin, and mining wastes, to mention only a few of an almost limitless number.

Soluble materials that infiltrate during periods of recharge retain much of their integrity as the contaminated mass migrates to points of discharge, such as wells, streams, springs, seeps, and drainage tile. Following another period of recharge, a second mass of contaminated water descends and moves laterally. Several distinct masses may exist in an aquifer at any instant in time, extending down a flow line and thus resembling a string of beads or a dripping faucet. A contaminated mass maintains a relatively uniform shape as it migrates through an aquifer because of laminar flow conditions and perhaps differences in specific gravity, temperature and viscosity between it and the native ground water.

PREVIOUS WORK

Although commonly alluded to, cyclic fluctuations of ground-water quality are rarely described in the literature. In a number of reports, water-quality graphs actually indicate these fluctuations, but their significance is not described. Pettyjohn (1971a) reported that nitrate concentrations in a surficial aquifer in North Dakota increased substantially following periods of precipitation. He assumed that the nitrate was leached from decaying organic matter in the scores of small sloughs and potholes that dot the area. In a later report describing cyclic chloride fluctuations in a streamside aquifer contaminated by oil-field brine, Pettyjohn (1971b) suggested that salt is washed downward into the aquifer during each period of natural recharge. Pettyjohn (1975) also described chloride fluctuations in Alum Creek in central Ohio, a stream contaminated by the discharge of brine-rich ground water.

Walker (1973a, b) pointed out that nitrate in the soil is transported to surficial aquifers during late fall and early spring recharge periods. He also recognized that these highly mineralized masses of water migrate through aquifers as bulb-like slugs with relatively little mixing or dilution.

Toler and Pollock (1974) described the accumulation of deicing salt in the unsaturated zone along a major highway in northeastern Massachusetts. They pointed out that some of the salt was flushed deeper into the ground during the spring recharge period, reaching the water table during midsummer. This is clearly another example of cyclic change in ground-water quality.

Nightingale and Bianchi (1973) discussed the Leaky Acres ground-water recharge system at Fresno, California. Their graphs show rapid increases in specific conductance as well as nitrate and chloride concentrations at the beginning of each recharge period. Schmidt (in press) discussed water-quality variations in pumping wells and presented a rather thorough review of the literature. The reader is referred to Schmidt's report for additional information.

CYCLIC FLUCTUATIONS

This report describes the cyclic fluctuation of chloride in a streamside alluvial aquifer in a humid environment. Chloride-rich wastes were both spilled on the ground and placed in pits for disposal. Similar water pollution situations might occur due to waste disposal by land spreading, burial, use of disposal or holding ponds and lagoons, and perhaps even septic tanks, among

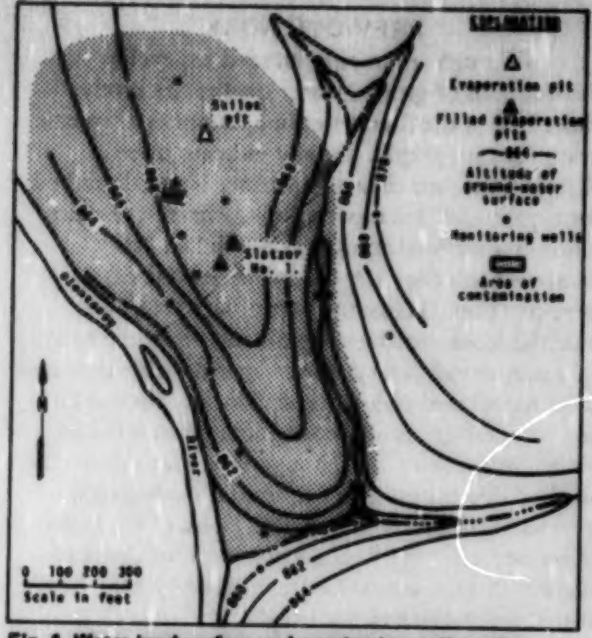

Fig. 1. Water-level surface and monitoring wells at the contaminated site.

several others. The example described herein may not be applicable to arid regions because of the commonly great thickness of the unsaturated zone and the limited amount of precipitation, soil moisture and ground-water recharge.

The major part of this report is based on data collected intermittently over the last 12 years, largely by The Ohio State University students, from an aquifer in central Ohio contaminated by oil-field brine (Shaw, 1966; Boster, 1967; Hulman, 1969; Lehr, 1969; Pettyjohn, 1971b, 1975; Kerzner, 1973). Similar phenomena, however, have been observed in many other areas throughout the country, particularly in humid and semiarid regions.

The contaminated site lies on the floodplain of the Olentangy River and is bounded on the east and south by small intermittent streams and on the west by the river. The land surface rises to the north (Figure 1). The alluvial deposits, consisting of sand, gravel, silt and clay, range from 15 to about 35 feet in thickness. The underlying bedrock is shale.

Since 1965, 23 test holes have been drilled in this area, ranging from 5 to 25 feet in depth. The water table lies from 1.5 to 5 feet below land surface and fluctuates a maximum of a foot or so throughout the year. Precipitation averages about 38 inches per year.

The site became contaminated during 1964-65 following the excavation of three brine holding ponds. One pit (Skiles) was used from July 1, 1964 to June 30, 1965; about 126,000 barrels of brine were placed in it. Two other pits (Slatzer number 1), also used at about the same time, received 110,000 barrels of brine. The impoundments, used only about a year, received at least 236,000 barrels of brine containing about 35,000 mg/l (milligrams per liter) of chloride.

The brine was spilled on the ground, but largely stored in the holding ponds. Fluids leaked from the ponds and soon contaminated the ground water. When samples were first collected from monitoring wells in July 1965, the aquifer, at least locally, contained more than 35,000 mg/l of chloride. At this time, the ponds were abandoned and the two Slatzer pits were filled with previously excavated materials that had formed surrounding embankments. During 1965-66 chloride concentrations in several wells, although variable with time, were in excess of 15,000 mg/l. By 1969 chloride concentrations in most of these wells were less than 2,000 mg/l, in 1972 were less than 1,000, and during a single sampling period in 1975, concentrations were less than 400 mg/l.

An interesting relationship becomes apparent when examining the areal extent of ground-water contamination with time. For example, the position of the 1,000-mg/l isochlor as it existed during selected months of 1965 and 1966, and the 500-mg/l isochlor during 1969, are shown in Figure 2. Note that the enclosed area during 1965-66 changes monthly but that the changes, even those from one month to the next, do not necessarily encompass smaller areas. This suggests

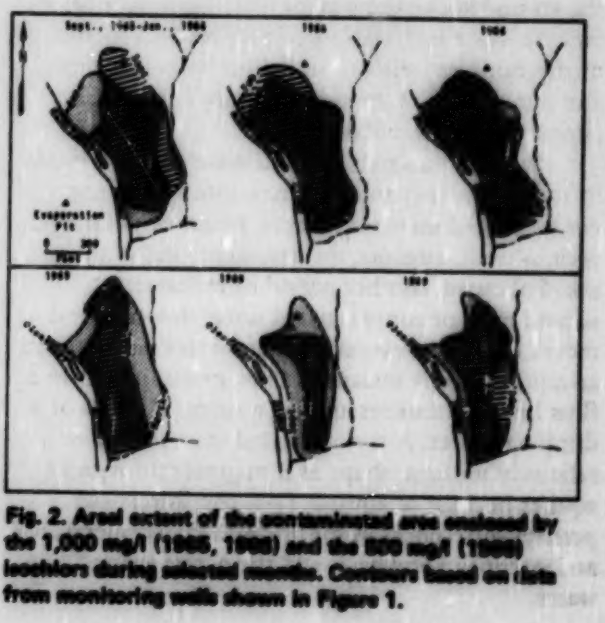

Fig. 2. Areal extent of the contaminated area enclosed by the 1,000 mg/l (1965, 1966) and the 500 mg/l (1969) isochlors during selected months. Contours based on data from monitoring wells shown in Figure 1.

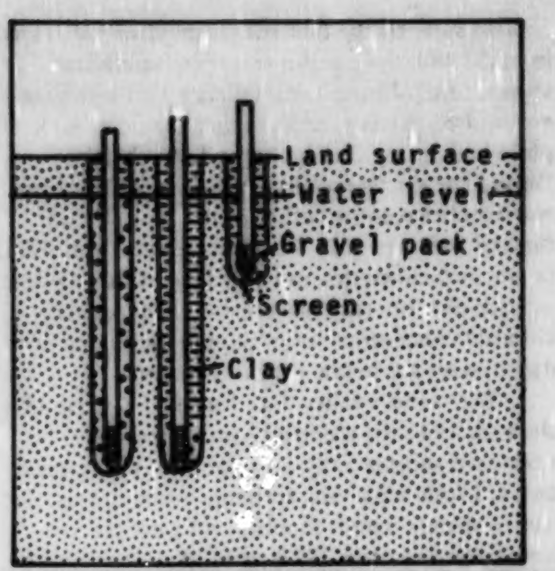

Fig. 3. Completion details of three closely-spaced monitoring wells.

that a linear relationship between chloride concentration and time (flushing rate) does not exist. For example, the area in September 1965 is larger than the January 1966 area indicating a natural flushing effect on the reservoir during the four-month period. But when the size of the January 1966 area is compared with the area that existed in February, and the March and April 1966 areas are compared, it is apparent that the contaminated zone has increased in size. Significantly, both February and April were wetter than previous months.

An examination of maps for 1969 shows a similar phenomenon (Figure 2). Monthly decreases in the size of the contaminated area are readily explainable as resulting from less rainfall and natural cleansing of the aquifer. It can be seen then, that in this area, both the size of the contaminated area and the chloride concentrations are functions of precipitation.

Of particular importance in the monitoring of this site are three adjacent wells, one screened at a depth of 9 feet and another at 23 feet, while a third is gravel-packed through much of its length and receives water from the entire aquifer (Figure 3). The location of these three wells are shown in Figure 1 at the position marked A. It is assumed that the first two wells represent the quality that exists at depths of 9 feet and 23 feet, respectively, and that the well gravel-packed throughout the entire thickness of the aquifer provides a composite sample of the reservoir. It is also assumed that when the composite well has a higher concentration than both the deeper and shallower wells, the most highly

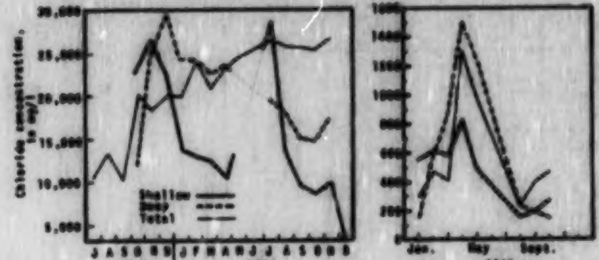

Fig. 4. Fluctuation of chloride content in closely-spaced wells of different depths during 1965, 1966 and 1969.

mineralized water lies between 9 and 23 feet.

The chloride fluctuations in the three wells during 1965, 1966, and 1969 are shown in Figure 4. Notice that at certain times the highest concentrations occur at the shallowest depth, at other times at the greatest depth, and at still others the greatest concentration must lie somewhere in the middle of the aquifer. The only means for accounting for this phenomenon is intermittent reintroduction of the contaminant, which is puzzling in view of the fact that oil-field activities ceased before any of the samples were collected.

Another technique illustrating the temporal-vertical distribution of chloride in the aquifer is shown in Figures 5 and 6. These illustrations are based on monthly data obtained from the three adjacent wells. Concentrations at depths of 9 and 23 feet were measured. Interpretation of the chloride distribution between these points was based on data from the fully penetrating well. In October and November 1965 the highest chloride concentrations were present at a shallow

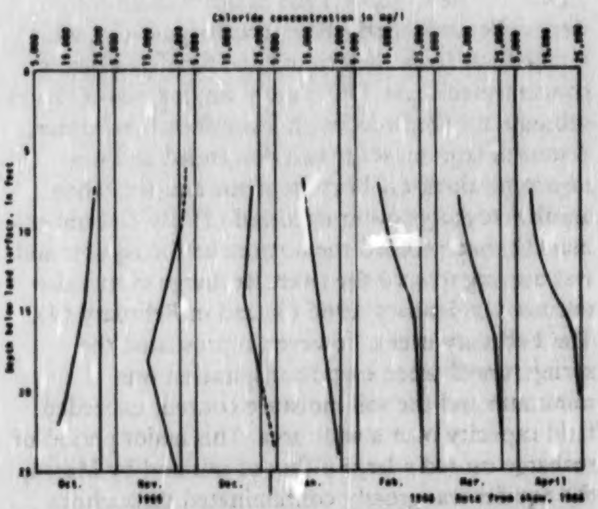

Fig. 5. Vertical distribution of chloride at the contaminated site from October 1965 to April 1966.

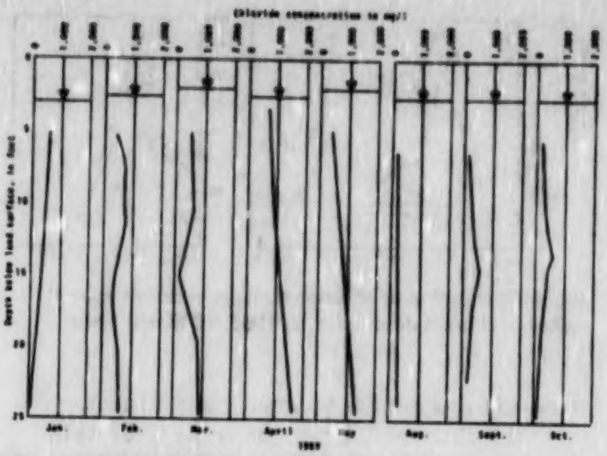

Fig. 6. Vertical distribution of chloride at the contaminated site during 1969.

depth, but during December 1965 through April 1966 the highest concentrations were near the bottom of the aquifer (Figure 5). Furthermore, in November and December, water in the middle of the aquifer was less mineralized than that above or below.

Although greatly reduced, in January 1969 the largest concentration of chloride was at the shallowest depth, but the situation was reversed during April and May (Figure 6). During February and March the central part of the aquifer was less mineralized than adjacent parts. By August there was only a slight chloride increase with depth, but in September and October the greatest concentrations again appeared in the central part of the aquifer.

The chloride fluctuations that occurred during 1965-66 and 1969 are shown schematically in Figure 7. The October 1965 samples apparently were collected shortly after a recharge event, which leached salt from the ground and formed a highly concentrated mass. This slowly sinking mass (1) was subsequently replaced with less mineralized water. A month later mass (1) had descended and was migrating along the bottom of the aquifer, when another recharge event occurred (2). By December, mass (2) had reached the bottom of the aquifer and was moving toward the river. Recharge events also occurred in January 1966 (3) and in February (4). The February event, however, represented the spring runoff when evapotranspiration was minimum and the soil moisture content exceeded field capacity over a wide area. This major period of recharge caused a large influx of salt and by March, the aquifer was grossly contaminated throughout a wide area. This mass eventually discharged into the river.

In spite of the fact that brine disposal ceased in mid-1965, the aquifer was recontaminated several times during 1969 (Figure 7). Following an established pattern, small recharge events took place in January, February, and March 1969. During the spring recharge period in April, there was another massive influx of brine as salt was flushed from the soil. This mass slowly diminished in size and concentration as it flowed through the aquifer towards and into the river. By August chloride concentrations, although increasing with depth, were far below previous months.

The water-quality data for 1965-66 and 1969 show that (1) the chloride concentrations decreased substantially over the four-year period but, (2) following each recharge event the aquifer was recontaminated. Similar cyclic events have been recorded from 1970 to 1975.

This study indicates that water-soluble substances that lie on the land surface or at a shallow depth may be intermittently introduced into a shallow aquifer for many years. The introduction of these contaminants is dependent upon the chemistry of the waste and soil and on the

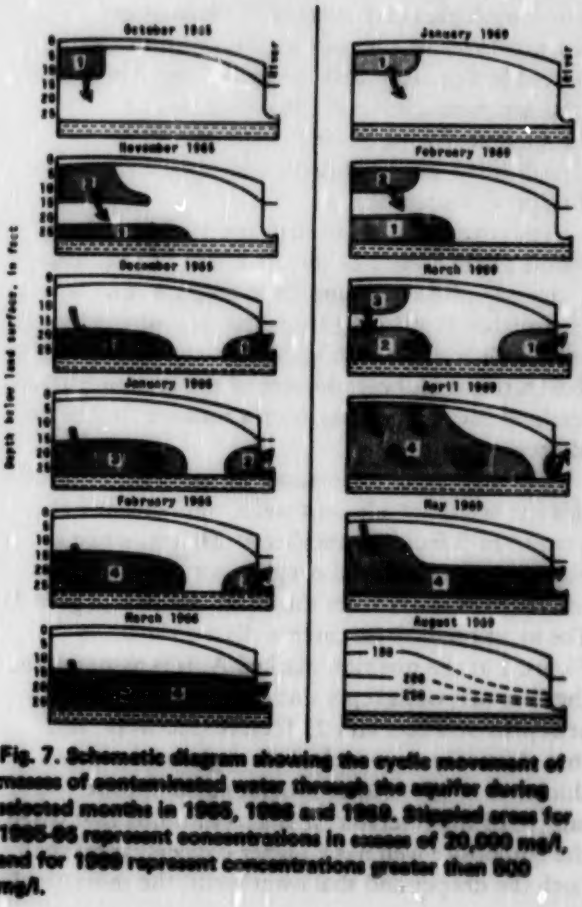

Fig. 7. Schematic diagram showing the cyclic movement of masses of contaminated water through the aquifer during selected months in 1965, 1966 and 1969. Stippled areas for 1965-66 represent concentrations in excess of 20,000 mg/l, and for 1969 represent concentrations greater than 500 mg/l.

frequency of recharge events, which are controlled by evapotranspiration, by the rate, duration, and intensity of precipitation and by soil-moisture conditions.

Throughout most of the year in humid and semi-arid regions, the quantity of water that infiltrates and the amount of contaminants that are washed into an aquifer are relatively small. On the other hand, during the spring recharge period, great quantities of contaminants may be flushed into the ground over wide areas. Therefore, the major influx of contaminants occurs on an annual basis, although minor recharge events may occur at any time.

When considering the possibility of cyclic fluctuations of ground-water quality, several points other than those alluded to here, must be considered:

(1) Even if there is a constant source of contamination, such as a septic tank, leaking transmission line, etc., more highly mineralized and cyclic masses could be generated through flushing during recharge events.

(2) In arid or semiarid regions where there is a thick unsaturated zone, the contaminated mass may never, or only rarely, reach the water table.

(3) In areas of high potential evapotranspiration, large concentrations of salts may lie near land surface. Since all of the available moisture is used by vegetation or evaporated, cyclic events would not be likely to occur. If, however, a rain of sufficient intensity and duration occurred, some of these salts could be washed downward.

(4) Even during steady-state conditions, such as induced infiltration from a contaminated river, the quality of the water may show cyclic fluctuations due to recharge between the river and the well or increased head in a holding pond due to the inflow of runoff, rain, or more waste.

(5) One must also consider aquifer stratigraphy, paying particular attention to horizontal and vertical permeability. Confining beds could cause a perched mass to migrate laterally, eventually to discharge through springs, seeps, or drainage tile. From the discharge point, it might again infiltrate and continue downgradient as an isolated mass. For example, Walker (1973a) described a case where mink farm wastes containing nitrate and bacteria flowed overland on glacial till more than a quarter mile before entering a creviced dolomite aquifer.

(6) Most commonly, cyclic fluctuations occur where water-soluble materials lie on the ground or in the ground above the water table and where the system is affected by direct infiltration. Cyclic effects may be masked where the contaminant must flow through confining beds.

(7) Movement of the contaminant in the saturated zone will be influenced by the difference in density between the contaminant and the receiving water. If the density of the contaminated mass is greater than the receiving water, and if the stratigraphy permits, the contaminated mass will move vertically downward and then horizontally.

(8) Wastes are not likely to infiltrate when the ground is frozen or when evapotranspiration is at its maximum. In a temperate or colder climate the greatest influx should occur during the nongrowing season, particularly in the fall and spring.

(9) Any soluble material may be transported deeper into the ground; enough rainfall events or prolonged rain may cause it to reach the water table.

CAVEAT EMPTOR

Oftentimes two or three water samples, collected over a period of a few years, are used to predict a flushing rate. If samples are collected at just the right time, we might assume a predictable rate, but we could be wrong. For example, an interesting relationship can be observed if we plot the chloride concentration of samples collected from a single well during the same month but in different years. Water from a well in the Olentangy River area discussed in the preceding pages had a chloride content of 35,750 mg/l in August 1965, 12,750 in August 1966 and only 380 mg/l in August 1969 (Figure 8). A straight line neatly fits all three points when plotted on semilog paper. If we assume the flushing trend will continue, the concentration should return to background (about 15 mg/l) by mid-1972. We might also plot chloride concentrations as they existed in the same well in April 1966 and 1969. Since it is easy to fit a straight line between only two points, the graph readily shows us that the area should be flushed clear by early 1973. Sampling and analysis showed, however, that the chloride content in April 1972 was not 15 mg/l but 1,900 mg/l; a recharge event had occurred just before the sample was collected.

Although seemingly ridiculous, we might use the same logic and work backward (both literally and figuratively) and fit a line through the concentrations of April 1969 (890 mg/l) and April

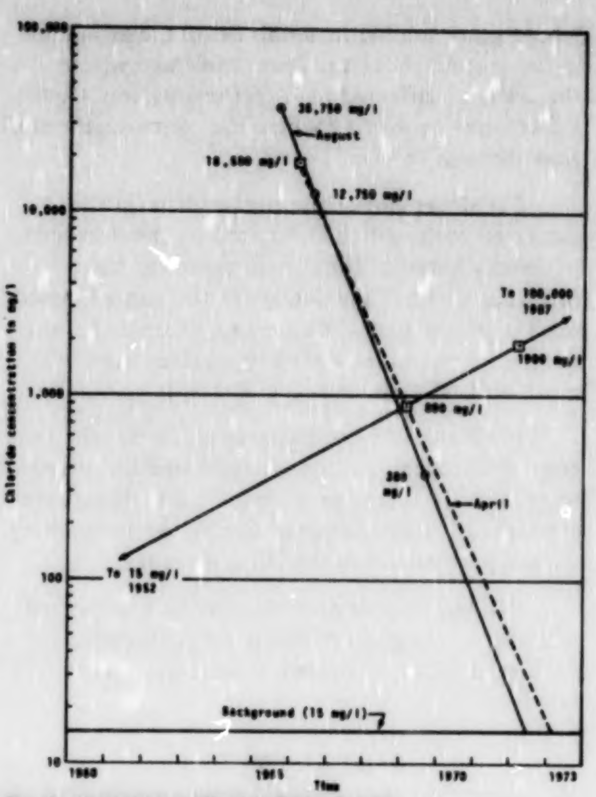

Fig. 8. Graph for predicting flushing rates.

1972 (1900 mg/l). In this case the graph indicates that the chloride concentration in the reservoir was at background in 1952, which it most certainly was, and that by 1987 the aquifer would contain a pure unadulterated brine, in spite of the fact that brine disposal ceased entirely in 1965!

A hypothetical cross section of an aquifer as it might have existed after two recharge events is shown in Figure 9a, along with 10 observation wells of different depths. A plan view of the area using only the "A" wells to develop isochlors is shown in Figure 9b. A similar map using only the "B" wells appears in Figure 9c, while Figure 9d shows isochlors determined from all the wells. Actual conditions, however, are shown in Figure 9e. If the monitoring wells were of different depths and closely spaced, the actual situation would have been evident.

The previous two illustrations show that we can make significant errors by using traditional methods. Only by employing knowledge of the aquifer system, determination of recharge events, and closely-spaced monitoring wells of different depths, coupled with common sense, can we adequately understand the movement of contaminants in ground water and thus design programs for monitoring ground-water quality in contaminated areas.

SUMMARY

Existing data indicate that in many situations, cyclic fluctuations of ground-water quality can occur and in fact may be common. These fluctuations are greatly influenced by the characteristics of the wastes, recharge events, and aquifer stratigraphy. Cyclic events can best be monitored by using a series of closely-spaced wells, each of which is screened opposite a small part of the aquifer and withdraws water from only that limited section. Moreover, samples should be collected from these wells at closely-spaced, regular intervals until the hydrologic nature of the site is recognized. Furthermore, we must not

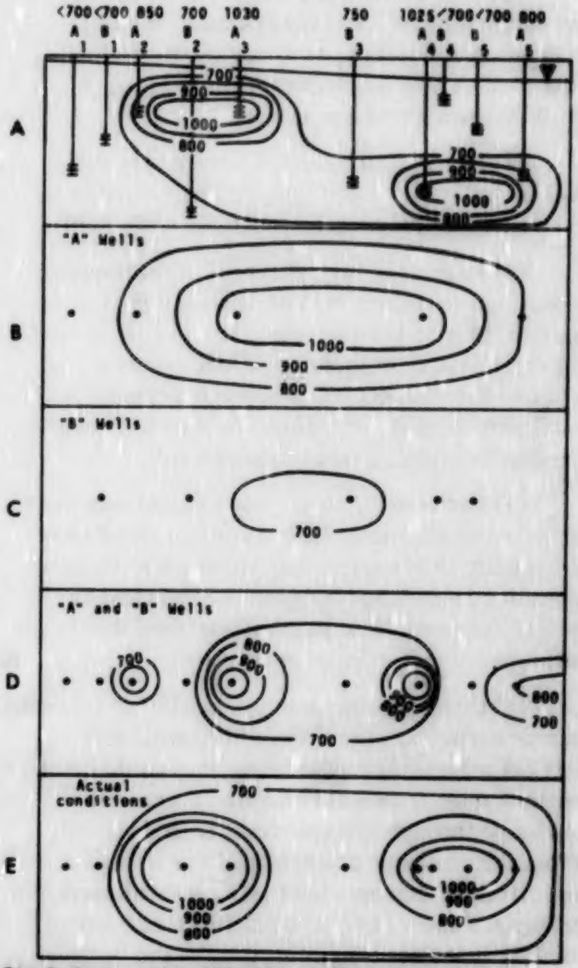

Fig. 9. Cross section (A) of an aquifer containing two contaminated masses and plan view interpretations of isochlors based on data from "A" wells (B), "B" wells (C), "A" and "B" wells (D), and actual conditions (E).

blithely pass over or ignore quality data that appear to be anomalous for they may tell us far more than the expected analysis.

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DISCUSSION

The following questions were answered by Wayne A. Pettyjohn after delivering his talk entitled "Monitoring Cyclic Fluctuations in Ground-Water Quality."

Q. *Isn't there a possibility that you have more than one source of chloride and that migration occurs under conditions of differential head under anisotropic conditions?*

A. Water from wells tapping till or alluvium in adjacent areas contains less than 15 mg/l of chloride. Likewise, chloride concentrations in the underlying shale and limestone are less than 25 mg/l. There are no septic tanks or any other waste disposal sites nearby except for the abandoned brine ponds. Road salting is not carried out on the country lane that crosses the site. Furthermore, the head in the alluvium is higher than that in the underlying shale or limestone. Therefore, brine disposal during 1964-65 provided the only source of chloride in the studied site.

Q. *Does the rate of flushing in your example have any correlation with the rate of ground-water movement or is it strictly related to recharge rates or other factors?*

A. The rate of flushing of this system depends largely on recharge events. Once the contaminants reach the aquifer, they move fairly quickly. If the system were not continually recontaminated, the aquifer would probably have been flushed clear of chloride years ago.

Q. *How long will it take for the brines to be completely flushed out of (1) the pit, (2) the zone of aeration, and (3) the aquifer?*

A. Until the brine is completely flushed from the zone of aeration, the aquifer will be intermittently recontaminated. The rate of flushing of the unsaturated zone is related to the frequency and magnitude of recharge, which, in turn, depends on the rate, duration, and intensity of rainfall, the season, soil-moisture conditions, grain size, and land use, among others. It is evident then that it is impossible to predict when the area will be flushed clear of the contaminant.

Q. *Was there a direct correlation between rainfall and recontamination?*

A. In general, there is a close correlation between rainfall and reintroduction of the contaminant. However, the amount of rain required to trigger an event depends on the antecedent soil-moisture conditions. For example, during the growing season more rain is required than during the early spring or fall. A rain that might cause a mass to form in the fall might have no effect during winter when the soil is frozen.

Q. *Are the rates of movement, both vertical and horizontal, reasonable to allow the slugs to move as postulated considering vertical and horizontal permeabilities and heads?*

A. Analysis of quality data permitted a rough estimate of

the rates of contaminant movement. Vertical migration appears to occur at an average rate of about .5 feet per day, while horizontal movement towards the Olentangy River is about 7 feet per day. Therefore, a mass would reach the bottom of the aquifer in 20 to 30 days and move from there to the river in about 25 days.

Q. ***If changes are due to recharge events, what is the best method to determine the background level?***
A. Background concentrations could be determined when evapotranspiration rates are the highest, when the soil is frozen, or several days after a recharge event. The time of frame, of course, is dependent on the vertical permeability and thickness of the unsaturated zone and the aquifer. One should not attempt to determine background concentrations in contaminated areas, but should limit sample collection to sites some distance upgradient or on the opposite side of a drain or divide.

Q. ***Why aren't the high concentrations of chloride initially flushed out? What mechanism caused the salt accumulation?***
A. At least 236,000 barrels of brine were disposed of at the site. As a result, the unsaturated zone, which consists largely of silt and clay, serves as a huge sink for sodium chloride, particularly in the vicinity of the pits. The alluvium contains an abundance of near vertical openings, such as dessication cracks, and animal and root borings, many of which are quite large. It is postulated that salty soil moisture diffuses from the surrounding contaminated but less permeable material into these larger openings, remaining there either as a salt crust (following evaporation) or as salty water, until it is flushed deeper into the ground during a recharge event.

In the unsaturated zone large pores drain quickly, while small pores drain much more slowly. Perhaps 90 percent of the water moves through 2 percent of the void space. However, gravity drainage of salty water from the small pores slowly moves downward, forming a highly mineralized zone near the water table. Much of this solution also is flushed into the aquifer during recharge. The zone is then slowly remineralized by descending gravity drainage from the small pores while the larger openings are reloaded by diffusion. When this stage is reached, a recharge event will cause another slug to form.

If the alluvium had consisted of coarse material, such as sand or gravel, the salt would probably have been flushed from the system fairly rapidly. Very likely the flushing rate could be increased by irrigation.

The Lycoming County, Pennsylvania, Sanitary Landfill: State-of-the-Art in Ground-Water Protection[a]

by M. Todd Giddings, Jr.[b]

ABSTRACT

A 120-acre sanitary landfill site is being developed in north-central Pennsylvania to serve a six-county area. The design of the site will prevent ground-water contamination by leachate from the refuse through the use of a 20-mil (.5 mm) PVC plastic liner which will lie between two protective layers of sand containing drainpipe networks.

Sampling of the composite ground-water flow from the underdrainpipe network in the sand blanket under the liner will provide sensitive monitoring of the liner performance. In the unlikely event of a leak, the pipe draining that area would be diverted to the leachate treatment facility, thus providing a backup leachate collection system. A thick (15 to 75 feet, 4.6 to 22.9 metres) glacial till deposit is present at the site and confines ground-water flow within the underlying shale bedrock. The low permeability of the till and the artesian head within the bedrock flow system provide additional protection against ground-water contamination.

Operation of the site will be by the area-fill method; the refuse will be deposited in 8-foot (2.4-metre) lifts up to a maximum height of 120 feet (36.6 metres). Based on an initial refuse deposit rate of 400 tons (363 metric tons) per day, the site is expected to have a 20-year life at a disposal cost of approximately $5.00 per ton.

Local residents who felt this project posed a severe threat to their wells strenuously opposed the project and appealed the State permit. The Pennsylvania Environmental Hearing Board upheld the permit; the decision was appealed to Commonwealth Court and was upheld. Residents remain unconvinced the design and site conditions will provide adequate protection and have petitioned the Pennsylvania Supreme Court to consider another appeal.

A baseline water-quality monitoring program has been undertaken to establish on-site conditions and a private-well sampling program will begin before the landfill is in operation. Till and bedrock monitoring wells at the site will be used to evaluate the performance of this landfill; analysis of the nearby private wells will be provided to the owners to demonstrate confidence in the design and the satisfactory operation of the ground-water protection measures.

INTRODUCTION

This 120-acre (48.6-hectare) landfill site is located on the Federal Allenwood Prison Camp property, approximately ten miles south of Williamsport, Pennsylvania. The Lycoming County Commissioners are developing this site to serve Lycoming County and the adjacent counties of Clinton, Columbia, Montour, Union, and part of Northumberland. All of these counties are located in the north-central part of the State where it has been necessary to develop a multi-county landfill operation in order for the sparsely populated areas to reach the minimum necessary solid waste volumes to make an environmentally-sound solid waste disposal site economically feasible. The site will serve an area of approximately 3000 square miles (7770 square kilometers) with a total population approaching 300,000 persons and, thus, will allow for the future development of a regional resource-recovery program.

The design of this site will prevent ground-water contamination by leachate from the refuse through the utilization of a tough, plastic membrane liner which will isolate the landfill from the ground-water flow systems. The leachate flow will be controlled and collected by the liner, subjected to biological and chemical treatment as required, and then will be recirculated back through the landfill to accelerate the stabilization of the refuse mass. The importance of this new design in managing the leachate problem is underscored by the Federal Appalachian Regional Commission's 1.2-million-dollar grant for construction of this sanitary landfill facility; this project will be administered by the U.S. Environmental Protection Agency (EPA).

HYDROGEOLOGIC FRAMEWORK

The site is located on the south slope of a hill within the Allenwood Prison Camp property in Brady Township, Lycoming County, and lies immediately west of U.S. Route 15. Topography at

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

[b] Principal, Todd Giddings and Associates, Consulting Hydrogeologists, 140 West Fairmount Avenue, State College, Pennsylvania 16801.

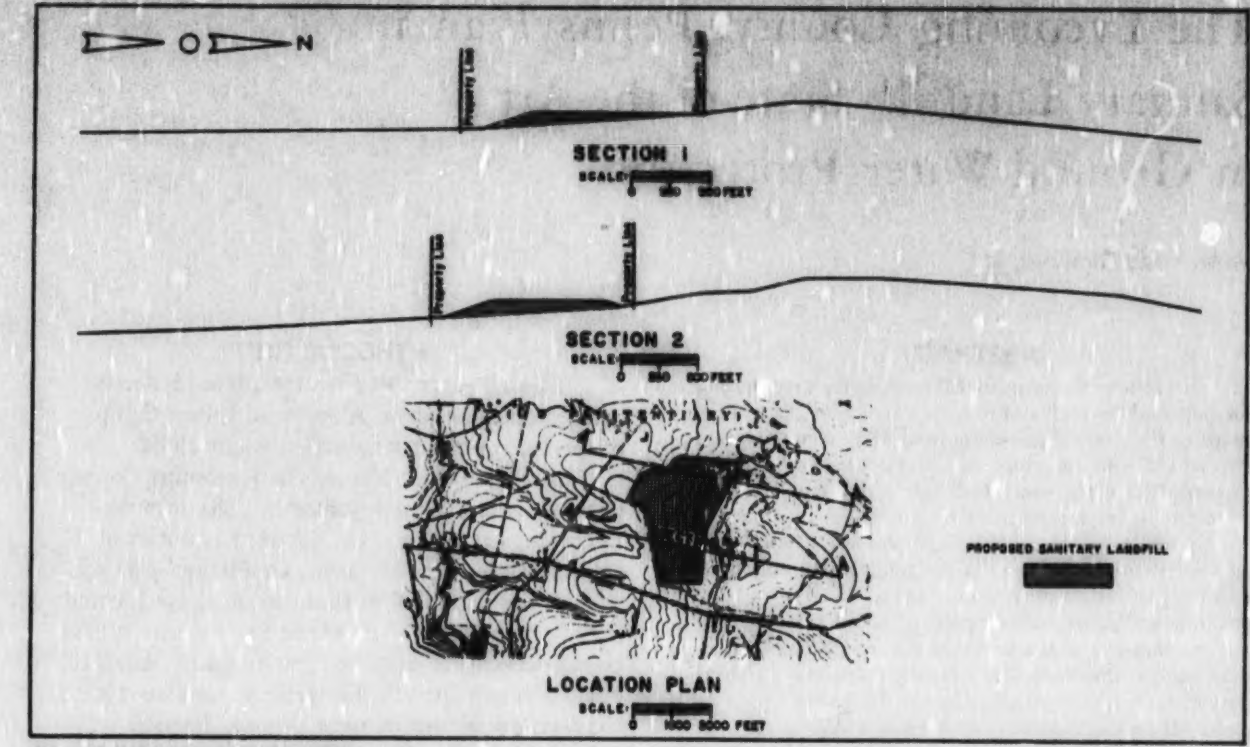

Fig. 1. Location plans and profiles.

the site is very gentle; near the upper or north edge of the site, slopes are approximately 13 percent but over much of the site the slopes are 5 percent or less (Figure 1). There is approximately 140 feet (42.7 metres) of relief on the site between the north and south boundaries (near the top and bottom of the hill). Within the site, there are several small gullies which drain the slope area and are up to several feet deep, carrying surface-water flow during heavy rainfall or snow-melt periods.

A detailed reconnaissance study of the site was conducted to map all pertinent surface features and to lay out the test drilling program. The site was flown to obtain vertical aerial photographs, and a photogrammetric topographic map was produced at a scale of 1 inch (2.54 centimeters) equals 100 feet (30.5 metres) and at a contour interval of 2 feet (.6 metre). This topographic map and aerial photographs taken at several different times were used to map features during the field study; this map was also used to present hydrogeologic data obtained during the subsurface exploration program.

A drilling program was conducted to determine the characteristics of the glacial till and the bedrock beneath the site. Eleven test borings were drilled through the compact, Illinoian glacial till, and cores were taken in the upper 5 feet (1.5 metres) of the underlying bedrock. Split-spoon samples were taken at 5-foot (1.5-metre) intervals within the till, and water levels were measured in all holes during and after drilling. The glacial till ranges in thickness from 15 to 75 feet (4.6 to 22.9 metres), with the area of thickest till located at the base of the hill along the south edge of the site. The non-sorted and non-stratified till material contains a wide range of sediment sizes including clay, silt, sand, pebbles, cobbles, and boulders, and is composed primarily of silt and clay material with scattered cobbles and boulders throughout. A detailed investigation of the soil series which have developed on this till was not undertaken because the construction of this site will involve excavation of all of the topsoil and several feet of subsoil in order to prepare a surface upon which to install the membrane liner.

The glacial till lies on bedrock which is dominantly a dark-gray shale with minor interbedded limestone and calcareous shale beds, with no evidence of solution openings. Eight of the 11 cores drilled at the bottom of each test boring encountered only dark-gray-brown shale. The other 3 holes encountered shale with some limestone interbeds. Strata beneath the site are quite flat, dipping only 10 degrees to the north.

Four ground-water quality monitoring wells, each 150 feet (45.7 metres) deep, were drilled around the perimeter of the site. All of these wells penetrated at least 100 feet (30.5 metres) of bedrock composed almost entirely of dark-gray shale. Cuttings from these air-rotary test holes indicated that some calcareous material was present in Wells 2, 3, and 4 (Figure 2); but because the bedrock is so nearly horizontal throughout the site area, and because the spacing of these wells around the perimeter of the site provides good control on the continuity and thickness of the predominantly shale bedrock beneath the glacial till cover, it was concluded that the limestone and calcareous shale material were present as discontinuous interbeds within the predominantly shale bedrock.

Bedrock crops out near the site, along the north edge where there is an abandoned shale quarry exposing flat-lying, interbedded sandstone and shale strata. No local folds or faults were observed in road cuts or shale quarries adjacent to the site.

A refraction seismograph was used to supplement the soil boring program by running traverses where the drill holes were furthest apart on the site. These data, in addition to the bedrock elevations from the test boring program, indicate that the bedrock surface beneath the proposed site has moderate relief apparently due to scouring and plucking by the Pleistocene-age ice sheet which overrode the hill immediately to the north. A bedrock-surface contour map was constructed from the test boring, water well, and seismographic survey data.

Drainage gullies are well developed throughout the site area and carry storm runoff and the discharges from several small springs within the glacial till. The watershed located above the north edge of the site is only a few acres in area and, thus, there is only a small amount of overland flow which will have to be diverted around the site. Soils throughout the site area contain a fairly consistent fragipan layer which causes shallow perching of water within the soil zone on this denser, clay-rich strata. Thus, there is a shallow perched ground-water flow system developed within the till, and lateral flow to small springs is common throughout the site.

Ground water within the shale bedrock beneath the till is confined by the overlying till blanket; all 4 of the air rotary test holes encountered ground water under artesian conditions, and the water levels in these wells all rose to within a few feet of the ground surface. The artesian heads ranged from approximately 9 feet (2.7 metres) at the north (uphill) edge of the site to more than 60 feet (18.3 metres) at the south (downhill) edge of the site. Water was first encountered in the bedrock at depths of 80 feet (24.4 metres) or more beneath the ground surface. The yields of the two wells along the north edge of the site, which are to be background water-quality monitoring points, were approximately 1 and 3 gallons per minute (.06 and .19 liters per second); the yields of the 2 downgradient monitoring wells along the southern edge of the site were approximately 20 and 30 gallons per minute (1.26 and 1.89 liters per second). Each of the 4 monitoring wells was cased and grouted at least 10 feet (3.1 metres) into solid bedrock so that no shallow ground-water flow contributed to the water levels or water-quality samples. Initially, 3-hour pumping tests were conducted on each of the monitoring wells to develop their yield and to obtain base-line ground-water quality samples. The ground-water quality samples were analyzed for a selected list of chemical parameters designed to establish base-line conditions with regard to these chemical parameters generally indicative of leachate contamination.

DESIGN FEATURES

Development of the site will involve the sequential placement of the 20-mil (.5-mm) PVC liner on 10 sections of the site (Figure 2). These sections, or "fields," will allow for the staged development of the site in order to match the current need for refuse disposal capacity. The first field to be excavated will provide a gently sloping floor which will then slope more steeply up to the terraced level of the second field to be developed. Eventually, the third and fourth fields will be developed along the west edge of the site to complete development of the entire western section of the landfill. A similar sequential development of terraced fields will allow development of the central section of the site, and finally the eastern section of the site will be filled.

The landfill will be operated by the area-fill method with the refuse being emplaced in 8-foot (2.4-metre) lifts and covered daily. The soil material excavated during preparation of each field will be stockpiled for use as daily, intermediate, and final cover. The maximum thickness of the refuse will be 120 feet (36.6 metres) where the gently sloping plateau reaches its maximum height above the slope of the hill. Thus, the configuration of the finished landfill will be a bench or a terrace on the side of the existing hill (see Figure 1).

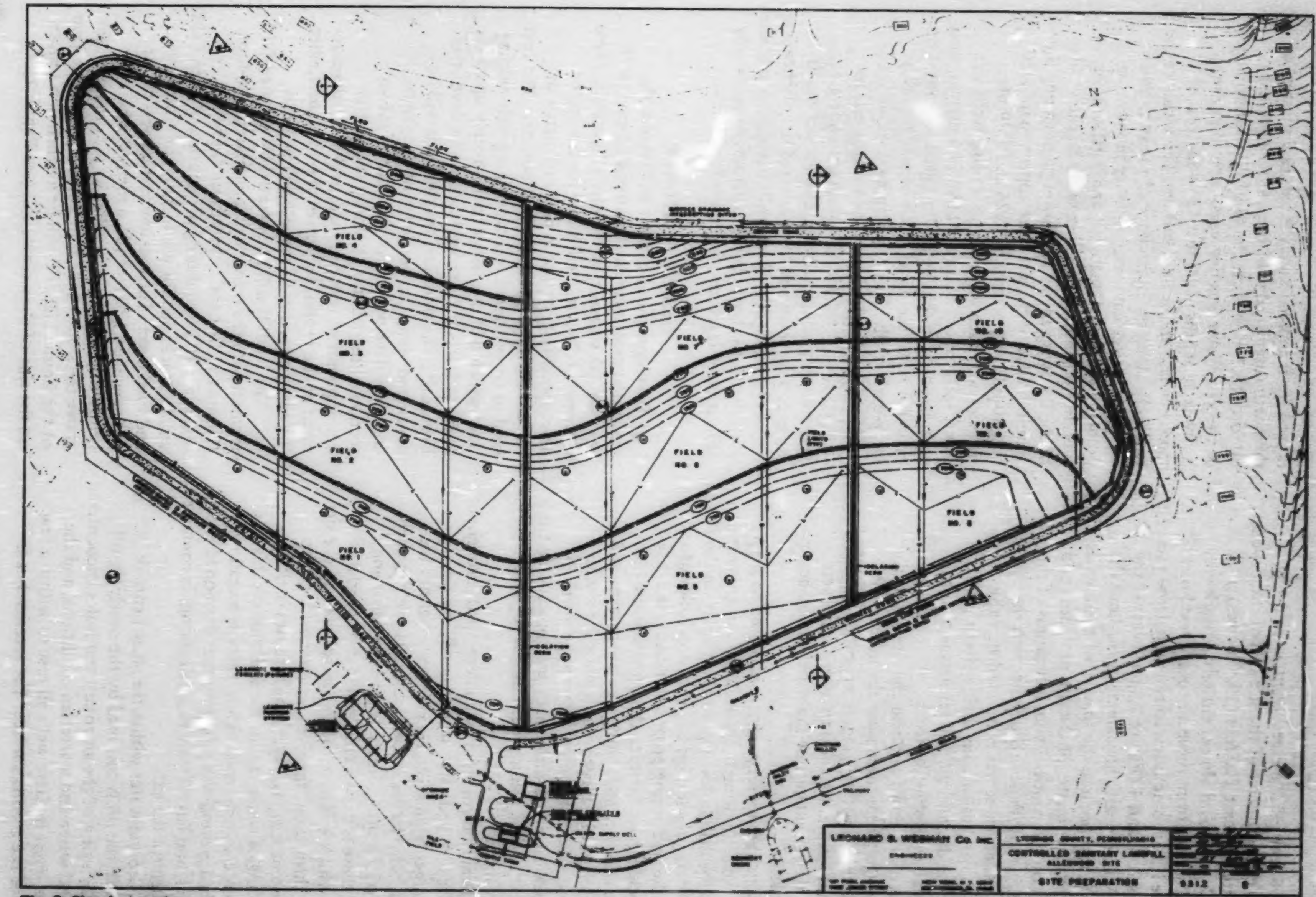

Fig. 2. Site design plan.

Reproduced from best available copy.

During construction of each of the fields, appropriate sedimentation and erosion control measures will be taken to control runoff from the site. As new fields are developed, previously used areas will receive final grading and seeding. The configuration and slopes of the finished landfill will be graded and seeded to provide pasture land for grazing, which is the present use of the landfill site.

The landfill is designed to receive municipal solid waste; chemical or hazardous industrial wastes will not be accepted at the site. The refuse will be spread and compacted by a landfill compactor machine in order to obtain an in-place refuse density of approximately 1000 pounds per cubic yard (600 kilograms per cubic metre).

The design of the ground-water protection measures includes several positive features not previously used in a full-scale landfill application. The leachate control system is a continuous multi-layer barrier, which separates the undisturbed glacial till and ground water from the refuse and leachate. From the bottom to the top, the barrier consists of the following layers: 12 inches (30.5 centimeters) of highly permeable sand (emplaced on the undisturbed till), a 20-mil (.5 millimeter) thick continuous sheet of impermeable PVC plastic, 6 inches (15.24 centimeters) of sand, and 12 inches (30.5 centimeters) of select clay soil (glacial till) (see Figure 3). Sets of perforated, horizontal drainpipes will be installed in gravel envelopes within the 6-inch (15.24-centimeter) sand layer overlying the plastic liner in order to carry leachate to the treatment and recirculation facility. There will be another set of horizontal drainpipes, also located in gravel envelopes, within the 12-inch (30.5-centimeter) sand layer beneath the liner, to provide an exit for ground-water seepage occurring beneath the liner. Much of the site is a ground-water discharge area for the shallow ground-water flow system located within the glacial till. Periodic water-level measurements made in plastic pipes installed in the 11 test borings located throughout the site indicate that throughout much of the site shallow water-table levels are within a few feet of the ground surface during all seasons of the year. This shallow ground-water flow system will be encountered during excavation and construction of each field, and the seepage from this flow system will drain from beneath the liner through the ground-water underdrain network (see Figure 2).

This ground-water underdrain network will provide a sensitive monitoring system to detect a leak in the liner. Should leachate be detected in the ground-water discharge from beneath the liner, the drainpipe set containing the contaminated water would be located by the sampling of each major, trunk drainpipe. The drainpipe set containing the contaminated water would be isolated, and that water would be conducted to the leachate treatment facility. Thus, this technique would provide a backup leachate collection system which would recapture the contaminated water should a leak occur. The possibility of a leak occurring is considered to be extremely unlikely, due to the protection of the liner by the sand layers, and the select soil cover. Also, the first lift (8 ft. or 2.4 m. thick) of refuse emplaced on the liner will not contain any coarse rubble or construction debris. Daily monitoring of the conductivity of the composite ground-water underdrain discharge will be utilized as an early warning monitoring system to evaluate the performance of the liner.

In addition to the engineered ground-water protection measures discussed above, there are also additional, naturally-occurring ground-water protection features at this site. The site is located within a shallow ground-water discharge zone within the glacial till. This flow system will oppose

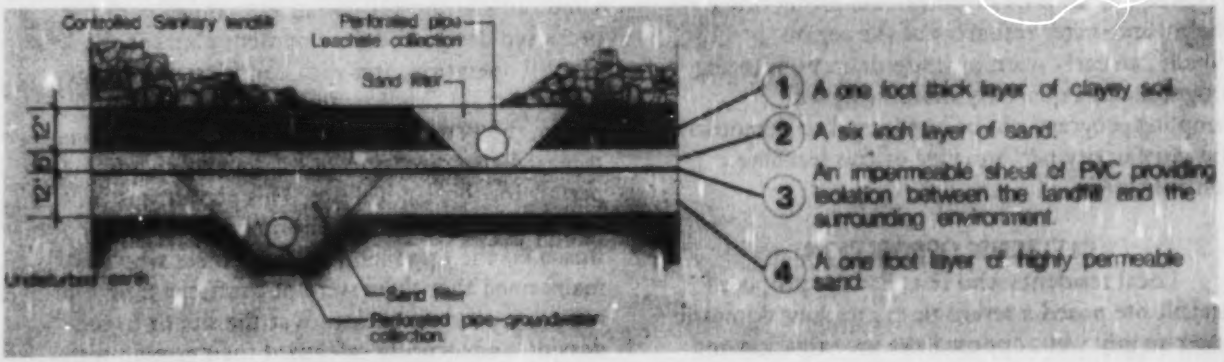
Controlled Sanitary landfill
Perforated pipe-
Leachate collection
Sand filter
12"
6"
12"
1 A one foot thick layer of clayey soil.
2 A six inch layer of sand.
3 An impermeable sheet of PVC providing isolation between the landfill and the surrounding environment.
4 A one foot layer of highly permeable sand.
Undisturbed earth
Sand filter
Perforated pipe-groundwater collection

Fig. 3. Design environmental protection measures.

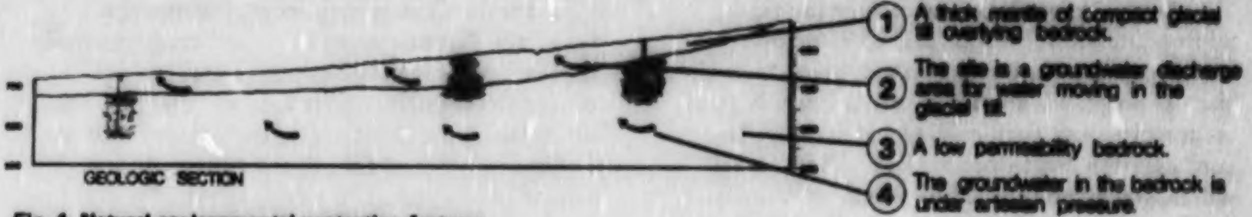
① A thick mantle of compact glacial till overlying bedrock.
② The site is a groundwater discharge area for water moving in the glacial till.
③ A low permeability bedrock.
④ The groundwater in the bedrock is under artesian pressure.
GEOLOGIC SECTION

Fig. 4. Natural environmental protection features.

any leachate trying to enter the till beneath the site. There will also exist a permeability contrast at the interface between the 12-inch (30.5-centimeter) sand underdrain blanket and the undisturbed glacial till. The contrast in permeability between this sand blanket and the undisturbed till will be approximately one million times; that is, the permeability of the sand will average one million times greater than permeability of the till.

Ground water within the shale bedrock underlying the till is under artesian pressure, with heads ranging from 10 feet (3.1 metres) to more than 60 feet (18.3 metres). This artesian pressure would prevent leachate from entering the bedrock flow system, should the leachate ever manage to enter the confining layer of glacial till above the bedrock. Finally, should leachate somehow enter the shale bedrock flow system and be detected in the downgradient monitoring wells, the low average permeability of this bedrock unit would allow for on-site extraction of the contaminated water before there is any degradation of the ground water in the region (see Figure 4).

Thus, there are 4 components of the multilayer barrier which are designed to separate the leachate from the ground water, and there are also 4 components to the naturally-occurring environmental protection system. The primary leachate collection system is designed with a backup collection system to recapture leachate within the ground-water underdrains. Further, several natural hydrogeologic features of the site are taken advantage of to provide additional protection of the ground-water resources of the region. Finally, an early warning underdrain monitoring program and a quarterly monitoring well sampling program will be conducted to demonstrate the satisfactory performance of this leachate control system.

CITIZENS' OPPOSITION

Local residents who felt that the proposed landfill site posed a severe threat to their domestic water-supply wells opposed the investigation and development of this site from the earliest phases of the project. Meetings were held with citizens' groups, and the site investigation procedure, landfill design requirements, and ground-water monitoring system were explained in detail. Most of the citizens remained unconvinced that there were adequate safeguards either within the design of the site or within the hydrogeologic features underlying the site, and the citizens' group retained a hydrogeologic consultant to assist them in evaluating the ground-water pollution potential of this project. Reasoning that the citizens might be fearful of the project out of ignorance of the safeguards, they were invited to inspect all phases of the hydrogeologic investigation of the site and were provided copies of all preliminary and final site investigation reports and landfill design plans.

During one of their tours of the site, the citizens discovered a shallow depression which they said was a sinkhole in the early stages of formation. Cavernous limestone strata are present in the area adjacent to the site, and unusual ground-water flow conditions and sinkholes are prevalent in local knowledge and folklore. A detailed evaluation of this depression was undertaken; the backhoe excavation revealed parallel buried stone walls on the margin of the depression and pottery artifacts at an old floor elevation within the depression. Thus, it was concluded that the depression was an old root cellar or building foundation associated with a farmstead present on the site before it was acquired for federal use.

After a thorough review of the hydrogeologic conditions at the site and after evaluation of the proposed design and development plan for the landfill operation, and with consideration of the questions and concerns raised by the citizens' group, the Pennsylvania Department of Environmental Resources issued a solid waste disposal permit for this landfill site. This permit was appealed to the State Environmental Hearing Board by the opponents of the project who still maintained that there were not sufficient environmental protection measures at the site or in the design to adequately safeguard their ground-water supplies. It was also claimed that the Department of Environmental Resources had not evaluated the design concepts for the site in enough detail in order to provide assurances that the site would function as designed. Additional questions were raised concerning the strength and durability of the 20-mil (.5 millimeter) PVC liner as the key component of the leachate control system.

The Environmental Hearing Board held 10 days of hearings wherein numerous experts testified as to each aspect of the site conditions and landfill design. Because neither the citizens nor their consultants raised any significant questions which were not thoroughly addressed or evaluated by the site study or landfill design, the Environmental Hearing Board upheld the issuance of this permit without reservation. The decision of the Environmental Hearing Board set forth findings of fact and findings of law which stated that adequate subsurface investigations had been conducted, adequate environmental protection measures had been designed into the site, and the review process carried out for this application had been thorough and proper in the procedure of evaluating these data and concepts.

In spite of this strong decision, the citizens' group appealed the Environmental Hearing Board's decision to the Commonwealth Court of Pennsylvania. Because the county had anticipated the possibility of this appeal, they had put into the record of the Environmental Hearing Board proceedings a considerable amount of detail through the testimony of their technical witnesses. Thus, the complete and thorough documentation and record which had been developed was then reviewed by the Commonwealth Court. That court stated that their review of what they termed a voluminous record compelled them to conclude that the Environmental Hearing Board decision was supported by substantial evidence and that the citizens' group was not denied due process of law nor was the permit issued arbitrarily or capriciously. Therefore, 9 months after the Environmental Hearing Board decision, the Commonwealth Court affirmed that decision.

Based on the strength of these 2 court decisions, Lycoming County is now proceeding with the final design of this site and the preparation of bid documents for construction. The citizens' group has now filed a petition asking the Supreme Court of Pennsylvania to hear an appeal to the ruling of the Commonwealth Court. In view of the strength of the previous decisions, it is considered unlikely that the Supreme Court will consider that appeal.

WATER QUALITY MONITORING PROGRAM

During the 2 years within which the construction of the landfill has been delayed by the permit appeal and court action, a program has been conducted which involved the periodic sampling of the monitoring wells and periodic measurement of water-table levels throughout the site. The 4 monitoring wells have been pumped and sampled annually for a period of 4 years; this program has provided base-line ground-water quality data for evaluation of long-term quality trends. In addition to the 4 bedrock flow system monitoring wells which were constructed on-site during the subsurface investigation work, another bedrock flow system monitoring well will be constructed downgradient of the site, and 2 shallower downgradient monitoring wells will be completed within the glacial till flow system at the south margin of the site. Thus, in addition to the early warning monitoring system provided by sampling the composite, ground-water underdrain discharge, there will be 2 monitoring wells at the downgradient (south) margin of the site within the glacial till, and 3 wells at the downgradient margin of the site located within the shale bedrock. These 5 wells will provide a sensitive, early-warning monitoring system for use in evaluating the performance of the leachate control system.

In addition to the continued ground-water quality baseline monitoring program on-site and in addition to the completion of 3 additional wells of this monitoring program prior to operation of the site, a private well monitoring program has been initiated. There are approximately 40 homes with private water-supply wells located within ½ mile (.8 kilometer) of the landfill site and 6 other springs or wells at a greater distance from the site which represent well-known or unique features within the regional flow system (which includes a significant zone of carbonate strata). The private well ground-water quality monitoring program (at no cost to the homeowner) would be for the purpose of documenting the existing ground-water quality conditions at nearby homes prior to and during operation of the landfill. This would protect both the county and the homeowners in the event of a change in water-quality conditions. Lycoming County will benefit from this program through the "insurance policy" effect which would result from the documentation of the water-quality conditions at each of the wells. These water-quality data would allow a thorough evaluation of any claims alleging damages from the landfill operation to a domestic well. By providing each homeowner with a copy of the analytical results each time his well is sampled, this program would clearly demonstrate that the county cares that adjacent residents worry about their water quality and that the county has nothing to hide and has absolute confidence in the ground-water quality protective measures at their landfill site.

SUMMARY AND CONCLUSIONS

This particular project was selected from among more than 75 sanitary landfill projects in which the author has been involved because there are several unique aspects of this project which set it apart from the usual case-history example project.

1. The utilization of a plastic membrane liner for leachate collection and control at this site as part of a 4-component engineered barrier system represents the state-of-the-art in lined sanitary landfill design.

2. Backup leachate collection through diversion of the ground-water underdrains provides an additional level of protection in the unlikely event of liner failure.

3. The naturally-occurring ground-water protection features present at this site provide yet another level of protection against the degradation of ground water from leachate at the site.

4. The Federal Appalachian Regional Commission grant of 1.2 million dollars for the construction of this sanitary landfill emphasizes the importance of leachate control in this design.

5. This project has served as a test case wherein the completed specifications for the composition, installation, performance, and guarantee of the membrane liner to be installed at this site are presented in the EPA publication "Liners for Land Disposal Sites; an Assessment."

6. Site problems are overcome through the use of a membrane liner, and site benefits are utilized in the design of this landfill to increase the degree of ground-water protection and to improve the sensitivity of performance monitoring.

7. Questions and concerns raised by the citizens' group opposed to the development of this site were investigated and answered during the hydrogeologic investigation of the site and during the design of the landfill and development of the operational plans.

8. The landfill site permit issued by the Pennsylvania Department of Environmental Resources was appealed by the citizens' group. Ten days of technical testimony before the Environmental Hearing Board examined all aspects of the site conditions, landfill design, potential environmental impacts, ground-water protection measures, and monitoring systems. The permit issuance was upheld by the Environmental Hearing Board without reservation.

9. Anticipating another appeal of the Hearing Board decision, a thorough presentation of all aspects of the landfill site investigation and design comments was presented before the Environmental Hearing Board. The Commonwealth Court affirmed the issuance of the permit due to the substantial evidence supporting the decision on the initial appeal.

10. This project, in spite of its state-of-the-art design, primary and multiple backup ground-water protection measures, and an extremely sensitive ground-water quality monitoring system does not satisfy some opponents. However, the regional problem of solid waste disposal was too great for this project to be dropped in the face of unfounded claims that the ground-water resources in the area would be contaminated by leachate from this proposed landfill site.

11. At an initial refuse disposal volume of 400 tons (363 metric tons) per day, it will be necessary to develop 4 fields of the site in order to have a lifetime of approximately 10 years. The total unit operating and capital cost for the first 10 years operation is estimated to be $5.00 per ton, which compares very favorably with fees at other landfill sites which do not provide the same degree of ground-water protection.

ACKNOWLEDGEMENT

The author wishes to thank the Leonard S. Wegman Company for furnishing their site design drawings to illustrate this report.

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DISCUSSION

The following questions were answered by M. Todd Giddings, Jr. after delivering his talk entitled "The Lycoming County, Pennsylvania, Sanitary Landfill: State-of-the-Art in Ground-Water Protection."

Q. by Bruce Brott. *Were background monitoring wells installed in both the bedrock and till aquifers?*
A. Background monitoring wells were constructed only in the bedrock aquifer. Two downgradient monitoring wells will be constructed in the till aquifer, and one additional downgradient monitoring well will be constructed in the bedrock aquifer to provide a total of three downgradient monitoring wells in the bedrock flow system.

Q. by Jeffrey A. Gilman. *Was the regional ground-water flow direction taken into consideration in the design and placement of the monitoring wells?*
A. The regional ground-water flow directions in both the till and bedrock flow systems, as determined from the hydrogeologic site investigation, were used to determine the location for placing additional downgradient monitoring wells in both of these flow systems.

Q. by James Braithwaite. *In general, do you feel it would be prudent to delineate [illegible] aquifer and soils characteristics outside and adjacent to the site in order to evaluate the contingency of leachate escape from the collection measures?*
A. Existing geologic mapping in the area of the site was confirmed in detail by the hydrogeologic investigation conducted within the site boundary. A reconnaissance study was made of the surrounding area up to a distance of between ¼ and ½ mile from the site in order to confirm the consistency of aquifer and soils characteristics outside of the area of detailed study.

Q. by James Gibb. *Is there a monitoring network around the lagoon?*
A. The downgradient monitoring wells for the landfill site are also located downgradient from the leachate-holding lagoon and, thus, will serve to monitor its performance also.

Q. by John Quagliotti. *What type of lining does the leachate-holding lagoon have?*
A. This lagoon will be lined with a 30-mil PVC liner placed on a prepared sand layer.

Q. by Herman Bouwer. *What will be the cost of the 20-mil PVC material to be used to line the landfill site area?*
A. The installed cost of this material is estimated to be $2.50 per square yard.

Q. by Charles L. Kissman. *Did you investigate the possibility of degradation of the PVC liner by industrial wastes or chemical sludges if they are to be allowed in the landfill?*
A. Disposal of industrial wastes and chemical sludges will not be permitted at this landfill site. Since there is a possibility that a small quantity of an industrial waste or solvent could be disposed of in the landfill without the knowledge of the operator, a one-foot-thick clayey soil barrier has been provided on top of the six-inch sand layer which overlies the liner. When this soil becomes saturated, it will provide a capillary barrier and perch the flow of leachate immediately above this layer. The permeability of the six-inch sand layer and drainpipe network overlying the liner will prevent saturated conditions from existing and prevent the prolonged contact of concentrated leachate flow with the liner.

Q. by Alexander Zaporozec. *Why was the expensive leachate collection system designed when the natural conditions are favorable for the attenuation of leachate?*
A. The shallow water-table condition within the ground-water discharge zone at this site and the large quantity of refuse to be disposed of at the site would not allow for the adequate attenuation of large quantities of leachate within the glacial till underlying the site. The liner not only will serve to collect any leachate which is generated, but it will also prevent ground-water seepage from entering the refuse and generating large quantities of leachate. By minimizing the quantities of leachate which are generated, the over-all leachate treatment and site operation costs will be lower.

Q. by Robert D. Mutch. *What provisions have been made for leachate treatment and for leachate generation minimization?*
A. Leachate will be collected from the pipe network overlying the liner and flow to the leachate-holding lagoon where it will be treated by aeration from floating-fan aerators. The leachate quality may also require additional chemical treatment within the lagoon. Final cover on the landfill will be at least three to four feet of the native soil material from the site, which is a high-clay-content glacial till. This soil will be compacted, graded, and seeded in order to promote runoff and the evapotranspiration loss of water by plants.

Q. by James C. Warman. *Have you considered transpiration drying of the landfill as described by Dr. Fred Mols at the Second National Ground Water Quality Symposium?*
A. Leachate quantities and rates of generation were calculated by utilizing a water-balance method which took into account evapotranspiration of water from the surface of the landfill.

Q. by David W. Miller. *What other types of treatment will be used in addition to the lagoon for the treatment of leachate?*
A. When the quantity of leachate generated from the landfill exceeds the amount which can be recirculated back into the landfill, a treatment plant will be constructed on-site. The leachate will be recirculated to accelerate stabilization of the refuse mass and to cause the decrease in leachate potency so that treatment costs in this plant in order to meet stream discharge criteria will not be prohibitive.

Q. by G. J. Thabaraj. *What, if any, are the state and federal requirements for discharge of leachate to surface waters?*
A. Discharge criteria covering a range of chemical parameters similar to those established for sewage treatment plants have

been set forth by the Pennsylvania Department of Environmental Resources for the receiving stream adjacent to this site.

Q. by George Clark. *What mechanisms will you utilize to detect and prevent migration of methane gas?*
A. Gas vents will be constructed at a spacing of 200 feet throughout the refuse deposit area and will penetrate the entire depth of the fill. These vents will be constructed of rock-filled, perforated concrete pipe sections and will allow leachate perched within the fill to drain to the leachate pipe network in addition to allowing the escape of gas from within the refuse mass.

Q. by Kenneth Lustig. *What was the cost of the site study, landfill design, and well construction to date?*
A. The cost of the site study including test drilling and the monitoring wells, the design of the landfill site and preparation of construction documents, and the coordination time of the county staff and legal fees to defend the permit through two appeal proceedings is approximately $300,000.

Q. by Kenneth Lustig. *What will the annual operating cost be of the completed landfill?*
A. The annual operating cost will be approximately $600,000 per year or about $5 per ton of refuse deposited at the site.

Q. by Edgar Meiser. *How much money has the Environmental Protection Agency allocated in grant funds for the study, design, and development of this site, and how does this grant affect the anticipated $5 per ton disposal cost?*
A. A grant of $1,299,379 is being administered by the Environmental Protection Agency for this project. If these funds had not been available, the disposal cost at this site would probably be in excess of $6 per ton.

Q. by K. E. Childs. *What is the life span in years of the PVC liner; will the liner last longer than the site will produce leachate; and is there a fund set aside to correct a failure if it occurs?*
A. The 20-mil PVC liner is expected to last far in excess of 20 years at which time it is expected that the potency of any leachate which is produced will be minimal due to recirculation and treatment of the leachate. There will be an operating fund set aside for contingencies, which could be used to correct problems with the leachate control system.

Q. by K. E. Childs. *Is there any possibility of sinkhole development beneath the site which could cause a major liner failure?*
A. There is no possibility of sinkhole development beneath the site because there is more than 1800 feet of shale strata underlying the site. Cavernous limestone strata are present beneath the shale strata, but the formation of solution openings at that depth below the site would have no effect on the structural stability.

Q. by Walter Steingraber and Jack E. Scava. *What course of action will be followed in the event that the leachate collection pipes become clogged with iron oxyhydroxide precipitates, as experienced in the collection of leachate from the Army Creek Landfill site in Delaware?*
A. Iron will be removed from the treated leachate which will be recirculated into the landfill, and the soil layer overlying the leachate drain system will also serve as a filter to prevent precipitates generated within the fill from reaching the leachate collection pipes. Should the pipes become clogged, there is adequate permeability within the gravel envelopes and sand layer adjacent to the pipes to allow for free drainage of leachate to the collection sumps at the lower end of the landfill.

Q. by Orest Tokarsky. *Did the consultants hired by the citizens' action group agree with most of your conclusions?*
A. Yes. During the hearings held following the appeal of this permit, a geologist and an engineer hired by the citizens stated that they felt the site investigation was comprehensive and entirely adequate in scope, and that the over-all design of the landfill represented the current state-of-the-art.

The paper by Michael Apgar, and the paper by James Mang, Robert Stearns, and Dallas Weaver were not available for publication.

Ground-Water Chemical Quality Management by Artificial Recharge[a]

by Harry I. Nightingale and William C. Bianchi[b]

ABSTRACT

The effectiveness of basin ground-water recharge at the Leaky Acres Recharge Facility in Fresno, California for improving the regional ground-water quality was studied as 65,815,000 m^3 of high-quality surface water was recharged from 1971 through 1975. Observation wells at the facility showed some variability in chemical parameters associated with each recharge period. The long-term decreases in salinity could be described by a power law decay curve fitted by regression analysis.

Without a special network of observation wells outside the facility, scientific evaluation of the enclave of recharged water is not possible. A practical evaluation of water-quality changes is possible from producing water wells around the facility. However, the pumping well discharge-time variations, well depth, aquifer sequence, and prior use of surrounding land must be considered, since all of these factors affect the pumped-water quality and its seasonal variability. Recharge at Leaky Acres had noticeably decreased the ground-water salinity for a distance of up to 1.6 km in the direction of the regional ground-water movement.

INTRODUCTION

Ground-water chemical quality has frequently been degraded as a result of man's activities and use of this resource (Kaufman, 1974). The causes of degradation and alternative projects to prevent it or perhaps improve the chemical quality are unique for each ground-water basin (Cofer and Owen, 1975; Schmidt, 1975a).

Artificial ground-water recharge is an alternative procedure for consideration in ground-water quality management, especially when low-salt surface water is economically available and the geology is favorable. Even wastewater is currently being considered for recharge, but more research is needed before its utilization for recharge to aquifers for unlimited use becomes a reality (Baffa, 1975; Bouwer, 1974; Cofer and Owen, 1975).

In 1970, the City of Fresno, California, which is completely dependent on ground water for its domestic and industrial supply, initiated a basin-type artificial recharge project. Surface water from the Kings River is delivered at a cost of \$8.107 per 1000 m^3 (\$10 per ac-ft). This water is available because urbanization is replacing irrigated agriculture. The recharge facility, Leaky Acres, is located about 7.2 km upgradient from the main pumping depression which is apparent in the regional ground-water gradient due to the urban well field. During the first 5 years (1971 through 1975) 65,815,000 m^3 (17.388 billion gal) of low-salt water has been recharged at Leaky Acres. Various research investigations associated with the large basin-type recharge facility were conducted during this time (Nightingale and Bianchi, 1973; Bianchi and Lang, 1974; McCormick, 1975; Nightingale, 1975; Bianchi and Nightingale, 1975).

Evaluating the effectiveness of a basin recharge facility based on the regional ground-water quality is analogous in many respects to evaluating contamination enclaves from surface waste disposal sites (LeGrand, 1965; Kimmel and Braids, 1974; Palmquist and Sendlein, 1975).

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[a] Water Management Research, Western Region, Agricultural Research Service, U.S. Department of Agriculture, Fresno, California 93726. In cooperation with the City of Fresno Department of Public Works—Water Division. Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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We could not economically justify determining the detailed geological and ground-water hydrological factors needed to develop a theoretical shape of an enclave of high-quality recharged water for this specific site. We expected the ground-water chemical quality to improve without bacteriological changes in the ground water. Therefore, like at Leaky Acres, evaluating most ground-water quality changes, associated with large recharge facilities, will depend upon sampling the existing wells and the variations accompanying: multiaquifer extraction, well depths, different usage, and non-ideal areal distribution. This report considers five years of ground-water quality data associated with recharge at Leaky Acres as determined under the above conditions.

STUDY AREA AND METHODS

Geologically, the Leaky Acres Recharge Facility is located on a compound alluvial fan (Page and LeBlanc, 1969) composed of vertical and lateral heterogeneous layers of clay, silt, and sand. Water movement in the fan sediments is vertically impeded by the less permeable clay and silt horizons, as indicated by perched water, while lateral transmission is semiconfined specifically to the sandy horizons. The geology and stratigraphy in the upper 30 m at Leaky Acres has been well defined (Nightingale and Bianchi, 1973; Bianchi and Nightingale, 1975). Generally, there are three low permeability layers at 4, 11, and 20 m depths; each is about 1 to 3 m thick. In 1975, these layers limited the facility's average infiltration rate to 14.2 cm/day. The ground-water mound dissipates rapidly after each recharge period and is indicative of productive aquifers, as indicated by Schmidt (1975b). Well yields usually exceeded 126 l/sec (2,000 gpm); specific capacities averaged about 25 l/sec/m (120 gpm/ft) of drawdown and transmissivities averaged more than 2,480 m^2/day (200,000 gpd/ft).

Figure 1 presents the location of Leaky Acres and the location and use classification of peripheral wells used to evaluate the influence of recharge on the regional ground-water quality. Behnke and Haskell (1968) conducted a ground-water nitrate study in this area. The urban development south of E. Dakota Avenue (Figure 1; except for Fresno Air Terminal) consists of lot sizes of about 557 m^2 (6,000 sq ft), each with an individual septic tank disposal system so the nitrate contamination in the area was about 25 mg/l. Since north of E. Dakota Avenue is mostly abandoned farm land with few home septic tanks because of the clear-zone for the airport, nitrate levels were usually less than 10 mg/l.

The initial areal distribution of ground-water specific electrical conductivity (SEC) in the study area (Figure 2) was established in 1971 from data collected in the summer (May through August) before recharge. The rural area north of E. Dakota Avenue usually had SEC < 200 μmhos/cm.

The surface water being recharged at Leaky Acres is a low salinity, Ca-MgHCO$_3$ type water, delivered from the Kings River (29 km east of Fresno) to the north side of Leaky Acres by the Fresno Irrigation District's Gould Canal. Samples for analyses were collected at inlets to the uppermost basins on the same schedule as the ground-water sampling at Leaky Acres.

Changes in ground-water quality beneath Leaky Acres were observed by sampling ten 20.32 cm I.D. PVC plastic-cased water-quality wells midway between top and bottom of slotted section below the third slowly permeable layer. The frequency of sampling of the wells at Leaky Acres had decreased from twice a week in 1971, to once a week in 1972, and semimonthly in 1973 through 1975. Ground water from 25 to 30 producing peripheral wells of various use-classification was usually sampled semimonthly, unless they were temporarily out of service, (i.e. irrigation well during the winter), or monthly if they were

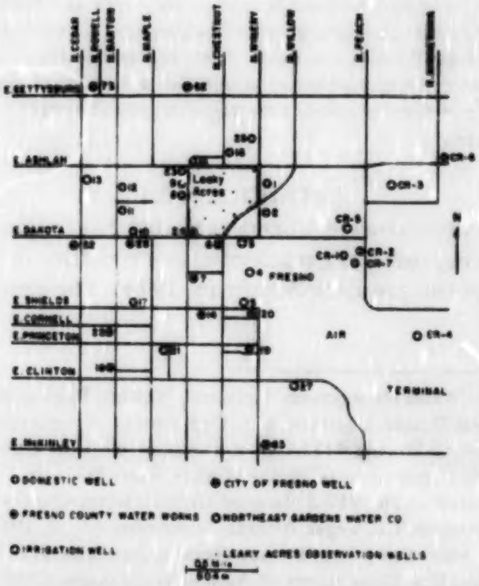

Fig. 1. Location of Leaky Acres Recharge Facility and peripheral ground-water quality monitoring water wells.

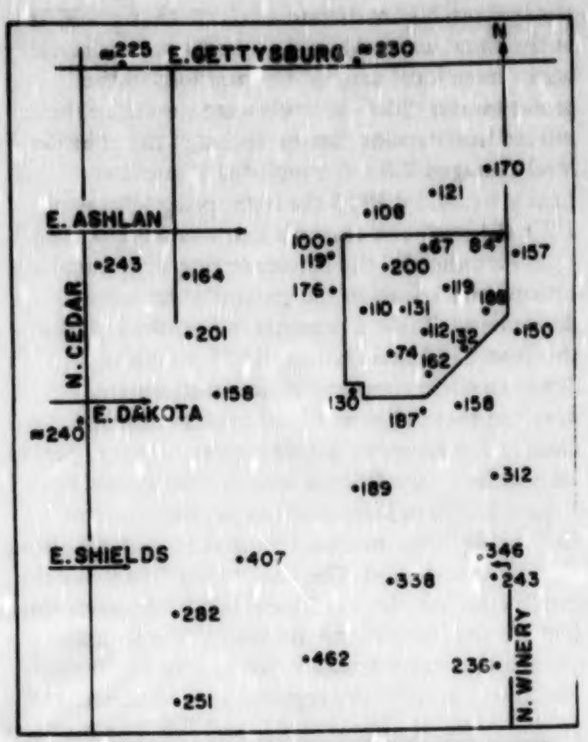

Fig. 2. Initial (May through August 1971) mean ground-water specific electrical conductivity, µmhos/cm, for study area.

fairly constant in quality. Water samples were immediately analyzed for their nitrate, chloride, and specific electrical conductivity (U.S. EPA, 1974). Ground-water turbidity and temperature changes were also measured. Major cations and anions were analyzed once yearly. For this study area, the total dissolved solids (TDS, mg/l) can be estimated accurately from the specific electrical conductivity (SEC, µmhos/cm) when in the 20 to 650 µmhos/cm range, by the regression equation: TDS = (SEC + 22.79)/1.62.

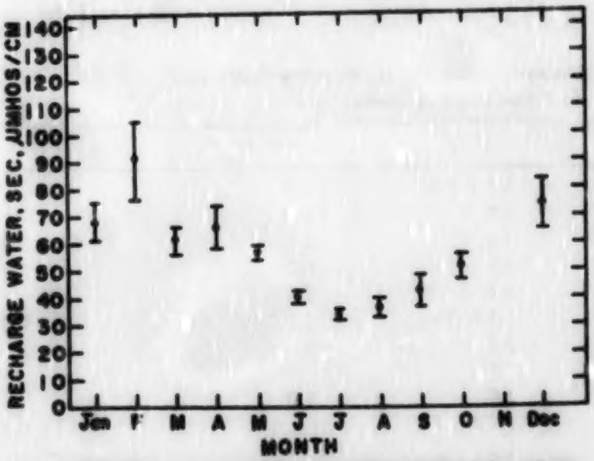

Fig. 3. Monthly mean specific electrical conductivity of the recharge water for 1971 through 1975 recharge periods. No recharge in November.

Table 1. Selected Canal Water Quality Parameters. Monthly Mean for 1971 Through 1975

Month	Cl^- mg/l	NO_3^- mg/l	FTU*	Temp. C	D.O.* mg/l
Jan.	2.51	2.77	13.0	8.8	8.00
Feb.	3.34	6.00	13.7	11.0	7.88
Mar.	1.58	1.21	8.7	13.0	9.80
Apr.	1.94	1.50	4.8	14.9	9.60
May	1.08	1.98	4.2	16.9	9.77
June	0.87	1.45	4.1	20.1	8.70
July	0.79	1.25	4.6	20.3	9.38
Aug.	0.72	0.95	2.8	19.8	9.00
Sep.	1.25	1.81	3.0	18.8	9.70
Oct.	0.78	1.28	3.6	16.6	9.34
Nov.	–	–	-No Recharge-	–	–
Dec.	2.65	1.78	4.5	10.0	8.00

* Turbidity in Formazin Turbidity Units.

* Dissolved oxygen.

RESULTS AND DISCUSSION

Canal Water Quality

The salinity of the canal water as evaluated by the SEC was usually between 30 and 70 µmhos/cm (Figure 3) during the recharge periods. The higher values resulted from runoff of some winter and spring storms. Table 1 presents the monthly mean chloride, nitrate, turbidity, temperature, and dissolved oxygen for 1971 through 1975. These data establish the excellent quality of the water before recharge.

Ground-Water Quality Trends — Beneath Leaky Acres

Figure 4 summarizes the ground-water SEC trends during five years of recharging 65,815,000 m^3 of high-quality water. Shortly after the beginning of each annual recharge period, there was some increase in ground-water SEC, but a new, low, SEC was attained by the end of the recharge period. The fluctuations in the Spring of 1973, shows the impact of turning the water on and off because of turbid canal water or shut-down periods in the western basins because of insect problems. The jump in SEC during August and September 1974, is in response to a gypsum treatment over tile lines used for water collection for a recharge well study.

Because of the variability (Figure 4) caused by irregular recharge periods, a power law decay curve was fitted to the monthly mean SEC data by regression analysis. The fit is satisfactory with the

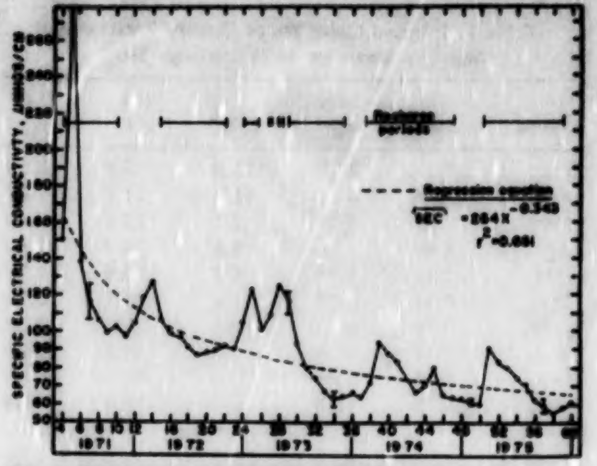

Fig. 4. Response in ground-water SEC beneath Leaky Acres as related to annual recharge periods. Each data point is the mean of all observation wells and sampling times for a month. In the regression decay curve equation, X is the number beginning with X = 1, for January 1971.

coefficient of determination, r^2 equal to 0.651. The regression equation would predict a ground-water SEC of 44 µmhos/cm by the December of 1980, 120 months after recharge started. This is reasonable if there is no recharge during December through February, when only water that is slightly higher in salinity is available. Essentially, the quality of the ground water beneath Leaky Acres is nearly in a steady-state equilibrium with the recharge-water quality.

Monthly mean ground-water nitrate levels beneath Leaky Acres are presented in Table 2. The nitrates in the soil profile before recharge were essentially leached out during the first recharge period. The slightly higher nitrate levels in the spring resulted partly from mineralization of organic matter during the winter dry period and partly from higher nitrate levels (5-15 mg NO_3^-/l) in the canal water when it contains surface runoff water from local storms. Fluctuations in the ground-water chloride levels were similar to the nitrate fluctuations. Before recharge the chloride level averaged 2.8 ± 0.3 mg/l for all wells at Leaky Acres, by 1975 the high spring level was 1.7 ± 0.1 mg/l and season's low was 0.7 ± 0.1 mg/l.

Periodically, the concentration of the major cations and anions in the ground water were determined. Table 3 presents the results only for the June 1971 and October 1975 samplings. Between these times the concentrations are between those shown. Fluctuations in these chemical parameters within a given recharge period are similar to the fluctuations in SEC shown in Figure 4. The milliequivalents per liter ratio of Ca^{2+} to Mg^{2+} has increased from 0.61 to 1.16 during the five-year period. The Ca^{2+} to Mg^{2+} ratio for the canal water for the 1975 data was 1.13, about the same as that for the ground water. The soluble silica concentration was within the 30 to 50 mg/l which is typical of the regional ground water. The pH has remained between 6.7 and 7.0. The dissolved oxygen content of the ground water averaged 8 to 9 mg/l at the start of a recharge period and decreased to 5 to 7 mg/l by the end of a recharge period.

Ground-Water Quality Trends – Beyond Leaky Acres

In evaluating the chemical data to determine the significance of recharge at Leaky Acres on the surrounding regional ground-water chemical quality, we must remember several factors: (a) The domestic wells are shallow, usually 30 to 40 m deep, open-bottom, and the volume of pumped water for house and yard use is small (as compared with public supply and irrigation wells), and the

Table 2. Ground-Water Nitrate, mg/l, Trends Beneath Leaky Acres. Monthly Mean, and for Some Months the Standard Deviation of Mean

Month	*1971*	*1972*	*1973*	*1974*	*1975*
Jan.	–	3.5	6.2 ± 0.9	4.9	–
Feb.	29.2 ± 8.3*	2.4	4.5	5.0	–
Mar.	–	6.6	3.2	5.1	4.0 ± 0.5
Apr.	–	3.2	3.2	4.2	–
May	–	3.5	2.8	2.5	–
June	15.3 ± 4.9	2.4 ± 0.5	2.3 ± 0.4	1.6 ± 0.1	1.7 ± 0.1
July	12.6	1.8	3.2	–	–
Aug.	5.3	3.9	2.5	–	–
Sept.	3.6	2.7	2.0	1.0	–
Oct.	3.2 ± 1.0	2.4 ± 0.8	1.7 ± 0.2	–	0.7 ± 0.1
Nov.	2.3	2.7	1.1	–	–
Dec.	3.5	8.1	1.2	–	1.2

* Before recharge.

Table 3. Change in the Mean and Standard Deviation of the Mean of Selected Water-Quality Parameters for All Wells at Leaky Acres Between 1971 and 1975

Quality Parameter mg/l	*June 2, 1971*	*Oct. 28, 1975*	*Percent Decrease in Mean*
Na^+	6.95 ± 1.32	2.77 ± 0.14	60.1
K^+	1.98 ± 0.20	0.82 ± 0.02	58.6
Ca^{2+}	5.97 ± 1.04	4.89 ± 0.26	18.1
Mg^{2+}	5.90 ± 0.60	2.55 ± 0.09	56.8
HCO_3^-	69.67 ± 5.23	36.12 ± 1.68	48.2
Cl^-	1.85 ± 0.21	0.70 ± 0.04	62.2
NO_3^-	12.6 ± 4.52	0.73 ± 0.09	94.2
SO_4^{2-}	38.6 ± 2.4	7.6 ± 1.1	80.3
SiO_2	48.0 ± 3.9	37.7 ± 3.4	21.4

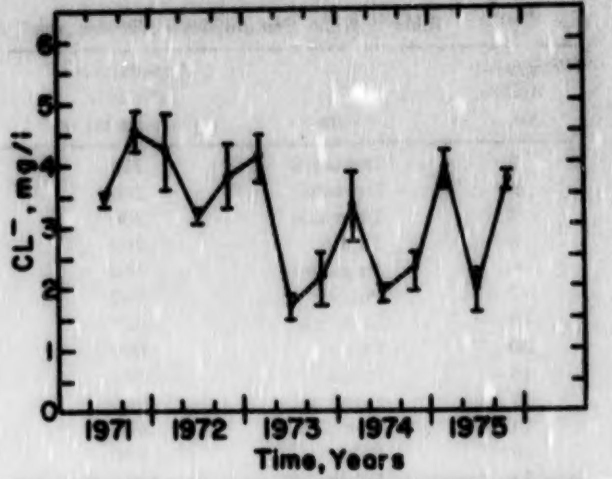

Fig. 5. Ground-water mean chloride concentration for each third of a year for peripheral well (PW) No. 7.

observed annual variability in water quality is low. For example, in 1974 periphery well PW-10 had an SEC coefficient of variability (Cv) of 2.9%. The domestic wells penetrate only the upper aquifers and can be polluted if they are in dense, septic tank areas, like PW-17, with a mean NO_3^- level of 45.3 mg/l with Cv of 7.5% in 1974, as compared with domestic well PW-11 (outside the dense septic tank area), with a mean 7.6 mg NO_3^-/l in 1974. (b) The urban area, which had only septic tanks, was converting to the Fresno Sewer System, and most connections were completed by the end of 1974. Thus, this pollution source will eventually be eliminated. (c) The public-supply wells are much deeper (75 to 100 m) and usually not perforated above 36 to 45 m. Fluctuations in pumped-water quality were observed in the public-supply wells in this area in response to seasonal use demand and drawdown. For example, PW-21 had a 1974 mean NO_3^- level of 32.3 mg/l with Cv of 27.1% and PW-7 had Cv of 80.3% variability.

The mean ground-water chloride concentration for the PW-7 (Figure 5), shows that during the summer months of maximum pumping, lower chloride levels (or SEC) were observed. This variability, which was not the same for all public-water wells complicates the long-term evaluation of effect of Leaky Acres' recharge. Regression analysis is probably the best practical way to make long-term evaluations. The seasonal cyclic variability is not great and for long-term effects could be disregarded.

Table 4 presents the results of power curve regression analysis for ground-water SEC for wells at various distances from Leaky Acres, as a function of time. The coefficient of determination for PW-6, 7, 14, and 21, in a south to southwest direction decreased with distance from Leaky Acres. This statistical method of describing trends in ground-water quality associated with basin-type recharge seems acceptable, considering expected life of the recharge facility, which is probably less than 30 years, and the unjustified costs of a scientific ground-water hydrological and quality study.

In the past ground-water nitrate levels in the public-supply wells of this area of high septic tank

Table 4. Ground-Water SEC Time-Trends Predictions for Peripheral Wells at Various Distances and Directions from Leaky Acres Based on Power Law Curve Regression Analysis of 1971 Through 1975 Data

						Estimated SEC, $\hat{y}$, µmbos/cm for Dec.					
PW NO.	*Approx. Distance m*	*Approx. Direction*	*No. of Obs.*	*Coef. of Deter., r^2*	*Regression Equation* $y = ax^b$*	*1971 x=12*	*1972 x=24*	*1973 x=36*	*1974 x=48*	*1975 x=60*	*1980 x=120*
2	200	SE	93	0.835	$\hat{y} = 356\,x^{-0.36}$	146	113	98	88	82	64
6	200	S	68	0.693	$\hat{y} = 389\,x^{-0.29}$	189	155	138	126	118	97
7	400	SSW	75	0.592	$\hat{y} = 346\,x^{-0.24}$	190	161	146	137	130	110
14	800	SSW	85	0.217	$\hat{y} = 387\,x^{-0.064}$	330	316	308	302	298	285
21	1200	SSW	104	0.106	$\hat{y} = 619\,x^{-0.13}$	448	409	388	374	363	332
11	800	W	83	0.404	$\hat{y} = 246\,x^{-0.12}$	183	168	160	154	150	138

* $\hat{y}$ = estimated specific electrical conductivity, µmhos/cm, after x-months from January 1971 when x = 1.

Table 5. Mean Ground-Water Nitrates, mg/l, for May through August, During Maximum Pumping

Peripheral Well No.	*Use*	*Approximate Distance from Basins, m*	*Nitrates, mg/l*				
			1971	*1972*	*1973*	*1974*	*1975*
2	Domestic	200	7.4	8.2	3.4	3.0	2.0
16	Domestic	200	4.3	2.3	2.5	4.1	2.6
8	Domestic	200	8.2	12.2	5.9	2.8	2.1
6	Public	200	14.2	15.1	9.1	7.4	6.3
4	Irrigation	400	34.1	OFF	30.9	OFF	32.8
7	Public	400	14.4	14.7	8.9	6.1	4.9
10	Domestic	600	9.9	12.1	11.0	10.9	7.5
20	Public	800	18.4	21.4	16.7	22.1	26.2
14	Public	800	30.0	30.1	31.5	22.8	35.8
17	Domestic	900	43.8	44.9	39.6	44.7	44.5
11	Domestic	800	6.0	7.1	5.0	7.4	6.6
19	Public	1200	18.7	23.5	19.9	28.8	28.4
21	Public	1200	35.1	36.9	31.2	33.7	31.9
18	Public	1600	23.4	25.1	23.3	27.6	24.5

density have received considerable attention. Table 5 presents summer (May through August) trends in mean ground-water nitrate levels, ranked according to approximate distance from the recharge basins. In general, only wells less than about 600 m have shown a nitrate decrease, and initially these were low in nitrates, because they were upgradient from the dense septic tank area. The nitrate levels of pumped water in the septic tank area have not greatly changed during this observation period. Unsaturated flow of poor quality water from the soil around the abandoned septic tanks probably will continue for several years because of necessary watering of lawns and gardens.

Figure 6 shows the areal distribution of the 1971-1975 percent decrease in ground-water salinity as evaluated by SEC from data for May through August (maximum pumping period). The decrease in SEC south of E. Shields Avenue is evident, but as shown in Table 5 the nitrate level apparently has not yet been noticeably reduced by recharge, probably because of the high septic tank density.

The Leaky Acres Ground-Water Recharge Facility demonstrates that ground-water quality can be improved or managed by basin-type recharge using high-quality water. Scientific evaluation of the enclave and movement of the recharged water is difficult without a specially built network of expensive observation wells and monitoring equipment. All recharge facilities should be monitored, which is possible if there are producing water wells around the facility. However, consideration must be given to their pumping capacity, delivery time schedule, depth, and zones of extraction. The historic and present land use around the recharge facility, and the stratification will affect the chemical quality of the pumped water and its seasonal variability.

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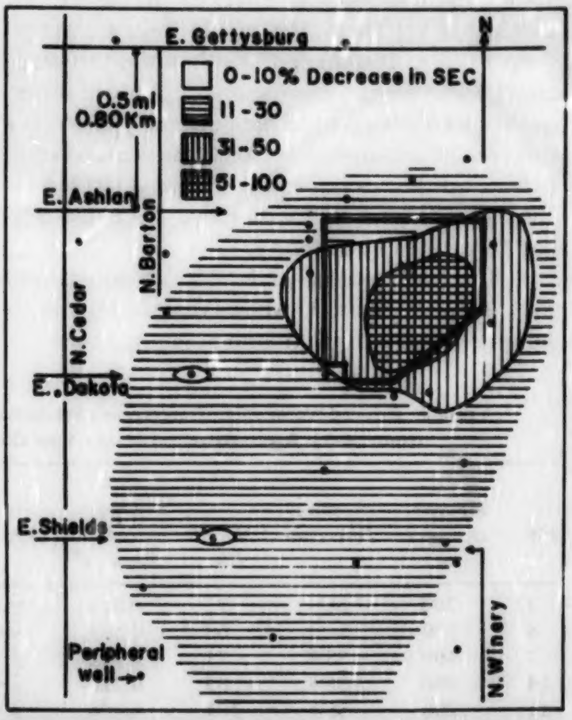

Fig. 6. Percent decrease in ground-water specific electrical conductivity (SEC) between 1971 and 1975, based on May through August data for these years.

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DISCUSSION

The following questions were answered by Harry I. Nightingale after delivering his talk entitled "Ground-Water Chemical Quality Management by Artificial Recharge."

Q. by Abe Kreitman. *Has the recharge capacity of the basins changed with time? If so, what is the nature of the change and what is the nature of the rehabilitative procedures employed?*

A. During the first five recharge periods the measured average infiltration rate for all basins has not statistically changed. The basin soils have been disked once during the winter drying period to mix the accumulation of organic material into the soil. We feel that disking should not be done unless necessary to prevent surface sealing due to accumulation of organic or inorganic sediments.

Q. by J. Brown. *Will the amount of recharge water available for the Leaky Acres project be affected by continued dry conditions in California or are quality criteria the only reasons for discontinuing recharge?*

A. During the dry year of 1976, water was made available for recharge through October even after the Fresno Irrigation District had to shut off surface-water deliveries to farmers on July 15. Perhaps this would indicate a higher-use priority, especially when the farmers also have ground water available for irrigation in water-short years. For the second part of the question, the historical chemical quality of the surface water available to Fresno for recharge would indicate that its quality would probably never be a criteria for discontinuing recharge.

Q. by Leonard Konikow. *Would you recommend the use of a deterministic model for studying artificial recharge problems?*

A. Yes, but only from the point of view that you have a real need and can justify the cost. There is no question that each recharge site is unique, and quantity and quality modeling is beneficial to our understanding recharge problems, but the problem of cost is difficult to justify.

Q. by K. E. Childs. *(a) Specifically what was the source of the recharge water? (b) Were the actual zones receiving recharge water monitored or were the monitoring wells in a different horizon?*

A. (a) The source of recharge water is the Kings River about 18 miles (29 km) to the east of Leaky Acres and is delivered by the Fresno Irrigation District's Gould Canal. (b) The zones receiving recharge water were actually monitored at Leaky Acres by observation wells perforated in the lower soil zones beneath the basins. Outside the bounds of Leaky Acres, domestic, public supply, and irrigation wells were monitored and they extracted water from many different sandy zones, which for wells further away from Leaky Acres, probably are not continuous with the zones monitored beneath the basins.

Q. by L. A. Swain. *Do you have any evidence that shows whether the better quality water recharged is vertically mixed throughout aquifer depth or floats on top of older ground water?*
A. The construction of the water wells monitored around Leaky Acres precludes the determination of vertical mixing based on chemical data of pumped water. The density difference between the native ground water and the recharged water is small.

Q. by Mike Kaczmarek. *In areas of short water supply where ground water is a major source of drinking water and municipal supply, where will you obtain recharge water?*
A. The management of the ground-water quality and quantity in a given area does demand the input of water of equal or better quality to maintain the quantity of water in storage, as well as an output of excess salt, be it to the ocean or a salt lake. The need for input by recharge should be first minimized by area-wide water conservation, especially keeping precipitation within the area, and maximum re-use of waste waters, then high quality surface water should be imported. If additional water is not economically available, then the people of the urban area should determine how they can continue to exist within their limited water supply and management capabilities.

Q. *What was the problem (need for recharge), and what does it cost?*
A. The problem in Fresno, California, is typical of urban areas in arid climates in that the water table has been falling as use has increased and septic tanks have resulted in a slow deterioration in ground-water quality in some areas. For the Leaky Acres Recharge Facility, and assuming 12,000 acre-feet per year of recharge, the total users costs are $16.98 per ac.-ft., which is composed of $4.56 per ac.-ft. for recharge operations, $10.00 per ac.-ft. for purchase of water (Bureau of Reclamation), and $2.42 per ac.-ft. for energy costs only to recover the water. For Leaky Acres there is a secondary benefit to the facility, namely benefits associated with Federal Airport clear zone land purchase contributions. If this benefit is considered, then the recharge costs are $3.53 instead of $4.56 per ac.-ft., which would reduce the total users' costs to $15.95 per ac.-ft.

The Dashte-Naz Ground-Water Barrier and Recharge Project[a]

by Dennis E. Williams[b]

ABSTRACT

The danger from salination poses a significant threat to development of ground-water resources for domestic or irrigation use in near-coastal environments throughout the world. The Dashte-Naz farm area near the Caspian Sea in northern Iran is no exception. Large-scale ground-water development in this region has been limited to wells south of the Caspian Sea near the base of the Alborz mountains where sediments are coarse and natural recharge high. In the northern regions, however, exploitation of the ground-water resources for agriculture and livestock purposes has proven fruitless with most of the high-yielding wells turning saline after only several irrigation pumping seasons. The source of the salination comes not from encroachment of Caspian Sea water but rather from sediments saturated with connate or fossil waters. A wide zone of dispersion separates the salinated aquifers from the fresh-water aquifers nearer the mountains.

The Dashte-Naz ground-water barrier and recharge project was conceived for purposes of overcoming the problem of exploitation of fresh ground-water reserves in the vicinity of saline-water fronts. The project presently under construction is a pilot research project on the practicality of using an injection well barrier for storing water underground in areas immediately adjacent to saline ground-water reservoirs. The results from this project will be used to design other water-supply projects in Iran not only along the Caspian Sea coast, but along the Persian Gulf as well.

INTRODUCTION

Salination of Aquifers

The word "contamination" has been used in the past to describe the condition of ground water which has been degraded in quality by the invasion of saline waters. However, contamination is generally associated with harmful chemicals or sewage, and is hazardous to public health. For technical continuity, therefore, it seems necessary to use a term explicit in meaning but not limited by definition as to source and concentration. The word "salinated" has been used throughout the literature for the past seven or eight years to describe this condition of aquifer degradation.

Salination of aquifers can result from several conditions depending upon the geologic and hydrologic factors active in the particular area. One of the most common occurrences resulting in aquifer degradation is from intrusion of saline water in near-coastal alluvial deposits. For example, along the coast of southern California, intrusion of salt water was first noticed in the early 1900's along the bay margin of Mission Valley in San Diego County. By the 1930's, the encroachment was detected as far north as Los Angeles, Ventura and Orange Counties. Since the 1940's, the natural seaward hydraulic gradient in many of the near-coastal communities has been reversed and control of salt-water intrusion is now the object of several

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

[b] Agro-Water Consulting Engineers, Tehran, Iran.

major projects along the coast of southern California. The problem, however, is not unique to southern California and is a common occurrence throughout all coastal areas in the world (the aquifers adjacent to the Caspian Sea or Iran being no exception).

The extent and degree of salination depends not only upon the geohydrologic factors affecting the vertical and lateral movement of the saline-water front, but also upon the extraction and recharge which is active in the area. The extraction or recharge may be natural or man-made, but whichever the case, the dynamic movement and shape of the saline-water front is greatly affected. To properly manage and control ground-water resources in close proximity to potential saline-water encroachment, one must first understand the ground-water hydraulics of flow governing this encroachment with a degree of confidence enabling proper prevention and control of all undesirable factors.

Definite progress has been and is being made regarding conservation and protection of ground-water resources near salinated zones in the northern coastal area of Iran near the Caspian Sea. In this area a project is presently underway in which an injection well barrier is being used to protect fresh ground-water supplies from salinated aquifers lying seaward to the north. The results from this study show that the well-known principle of a ground-water pressure ridge (as practiced extensively throughout the world for the past fifteen years), can and is being adopted for use in coastal areas of Iran (Figure 1).

History of Ground-Water Development in Iran

Since ancient times, Iran has been an agricultural country. Limited surface-water resources in most areas reflect the low annual average precipitation forcing farmers to develop ground-water resources to meet irrigation demands. Evidence of thousands of years of ground-water development can be seen by the numerous hand-dug shallow wells and ghanats scattered throughout the country.

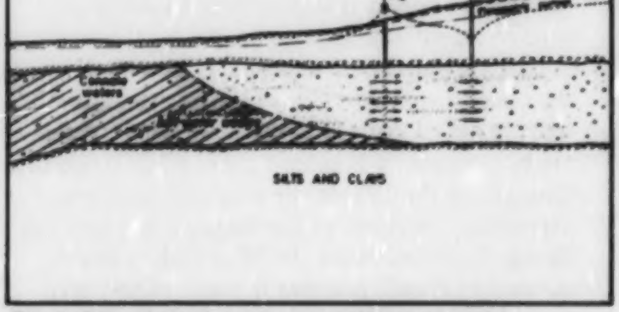

Fig. 1. Control of salt-water encroachment using an injection well barrier.

Ghanats represent a very old system of ground-water exploitation dating back thousands of years. Basically, they consist of subterranean tunnels dug mainly in alluvial fans with the upper portion of the tunnel contacting the water table. These tunnels act as drains on the phreatic aquifer providing a continuous flow of water, the amount and duration of flow dependent upon tunnel seepage losses and natural recharge. In recent years, decreasing phreatic water levels and ghanat maintenance problems have forced farmers to seek the untapped ground-water reserves lying in the deeper confined and semi-confined aquifers.

The advent of modern day drilling equipment has thus produced an explosion in the number of wells tapping the deeper aquifer zones with the result being an overdraft or mining condition occurring in many areas. Recent control by the government of Iran now requires conservation of all ground-water resources by application of modern planning methods on ground-water exploitation and use. The Dashte-Naz ground-water barrier and artificial recharge project reflects such planning.

THE DASHTE-NAZ AREA

Location

The area known as Dashte-Naz is located near the Caspian Sea in the northern part of Iran (see Figure 2). The total project area covers about 34,000 hectares with the Dashte-Naz farm area itself occupying about 3,000 hectares. The project area contains only one distinct topographic feature; that is, a plain with relatively smooth topographic highs and elevations ranging from −28 to +40 meters above the mean sea level of the Persian Gulf. In the southern portion of the project area, two alluvial fans exist formed by the accumulation of sediments from two large rivers (Tajan and Neka). A topographic depression exists between these alluvial fans as the result of the lack of sedimentation between the rivers.

The Need for Protection

The Dashte-Naz ground-water barrier initiated out of the necessity to supply irrigation water to

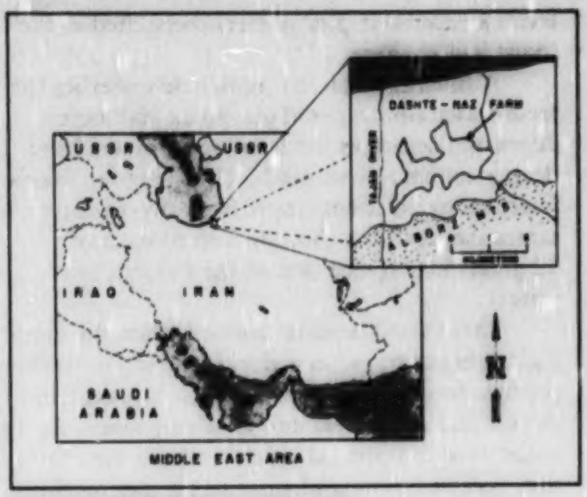

Fig. 2. General project location map.

the Dashte-Naz farm area. Originally, the project was conceived for the purpose of importing water from outside sources as it was thought that local ground-water resources existing in the aquifer systems within the farm boundaries were not fit for either irrigation or domestic purposes (early deep exploratory wells yielded electrical conductivities of 6000-7000 micromhos/cm). This latter theory was quickly disproved once results from precise exploratory cores were made available. Geohydrologic and geochemical test results show that fresh-water aquifers do exist in certain areas of the Dashte-Naz farm, namely in the southwest and northeast portions. However, exploitation of these fresh-water supplies is severely hindered by the ever-present threat of saline-water encroachment from the north.

Ground-Water Salinity Problems at Dashte-Naz

The two large alluvial fans in the project area are the result of deposition of sediments carried by the Tajan and Neka Rivers. The Tajan alluvial fan extends northward more than 20 km from the base of the Alborz mountains while the Neka alluvial fan, considerably smaller in areal extent, continues only 10 to 15 km northward from the base of the mountains. A definite topographic low point dividing these alluvial fans occurs in the vicinity of the Dashte-Naz farm. This low point coincides approximately with the mid-point between the two river systems (see Figure 3). The transgression and regression action of the ancestral Caspian Sea in Plio-Pleistocene times formed many interlayered alluvial and marine deposits in these alluvial fans (undisturbed core samples taken during the exploration phase show alluvial sediments interbedded with marine sands, silts and marls).

The salinity of ground water in the Dashte-Naz area generally increases from south to north reflecting the direction of alluviation. In the alluvial fans where the amount of natural recharge is high, finger-like lenses of fresh water protrude northward into areas of brackish-water aquifers. Specifically, fresh ground water occurs in abundant quantity near the Tajan and Neka Rivers where coarse sediments exist and transmissivities are high. Continuous circulation of fresh water in these areas has resulted in a flushing action over the past several million years cleansing the aquifers of the original connate waters. Away from the rivers, however, the magnitude of the natural flux decreases significantly resulting in brackish-water aquifers (Figure 4).

Geohydrologic and Geochemical Investigations

During the investigation phase of the project, over eighty deep exploratory boreholes were drilled for the purpose of differentiating fresh and salinated aquifers. Precise coring using the double-core barrel method combined with a complete suite of geophysical borehole logs produced the most concrete results regarding the geologic formations underlying the farm area. It soon became evident that fresh-water aquifers did in fact exist in the western and eastern portions of the farm near the alluvial fans of the Tajan and Neka Rivers. These fresh-water aquifers, however, are limited to the upper 100 meters or so of depth

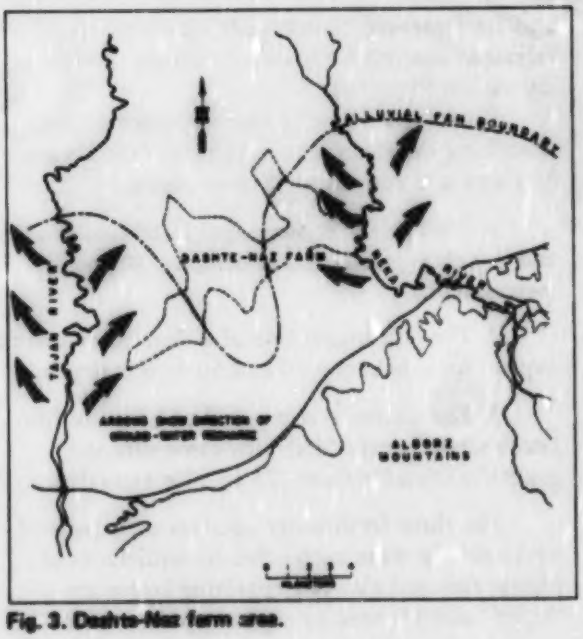

Fig. 3. Dashte-Naz farm area.

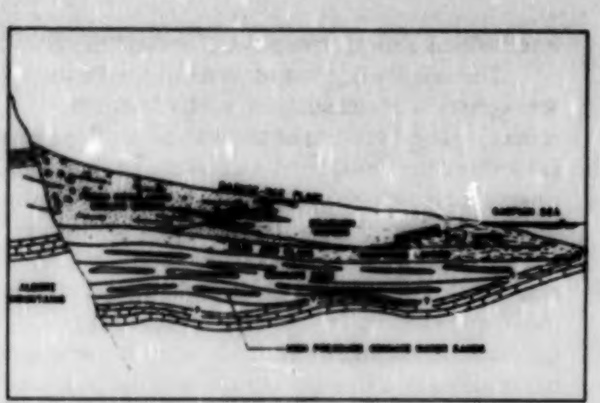

Fig. 4. Relation between fresh and salinated aquifers in Dashte-Naz.

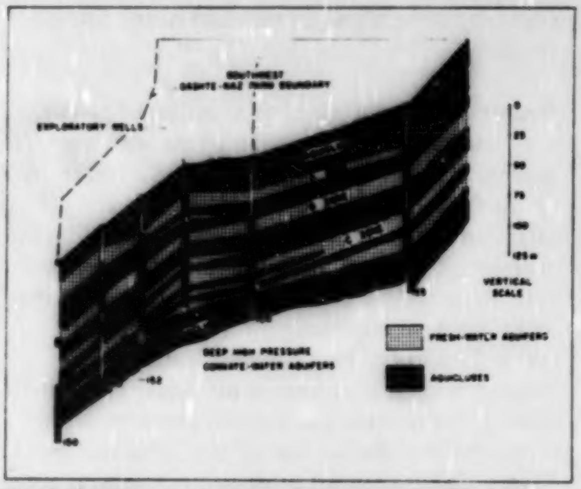

Fig. 5. Geologic cross section through upper fresh-water aquifers in the southwest farm area.

and are separated from deeper high-pressure salinated aquifers by thick silt and clay confining layers (see Figure 5).

A geologic "type" section was developed describing the fresh-water aquifers in the Dashte-Naz area and consisting of three zones:

1. *The A zone* is composed of alluvial sands and gravels typically found between 30 and 40 meters depth.

2. *The B zone* consists of a clean marine sand deposit lying between 50 and 60 meters depth.

3. *The C zone* is composed of layers of fine beach sands interbedded with some silts and generally found between 75 to 90 meters depth.

The three fresh-water aquifers are separated hydraulically from each other by aquicludes of plastic clay and silt. The confining layers are "tight" as evidenced by separate piezometric levels measured in piezometers completed in the individual aquifers.

A thick silt and clay aquiclude underlies the fresh-water aquifers at about 90 or 100 meters depth and separates the fresh upper layers from deeper connate-water sands. The pressure in these lower salinated zones ranges from two to three atmospheres with a chloride iron content of 12 grams/liter (twice that of the Caspian Sea water).

These lower aquifers formed when the overburden began to increase during the sedimentation process, forcing the salty waters to migrate from the silt and clay layers into adjoining sands. As the sands were compressed much less than the clays, they received more and more saline waters with internal pressures increasing as the overburden became thicker. The end result is a formation containing "pockets" of high-pressure saline-water sands hydraulically isolated from the fresh-circulating ground water.

A general geohydrological model of the area was developed after results from the exploratory drilling and testing became apparent. A two-layered model was adopted: the upper layer (above 100 meters) consisting of the fresh circulating ground water; and the lower layer (below 130

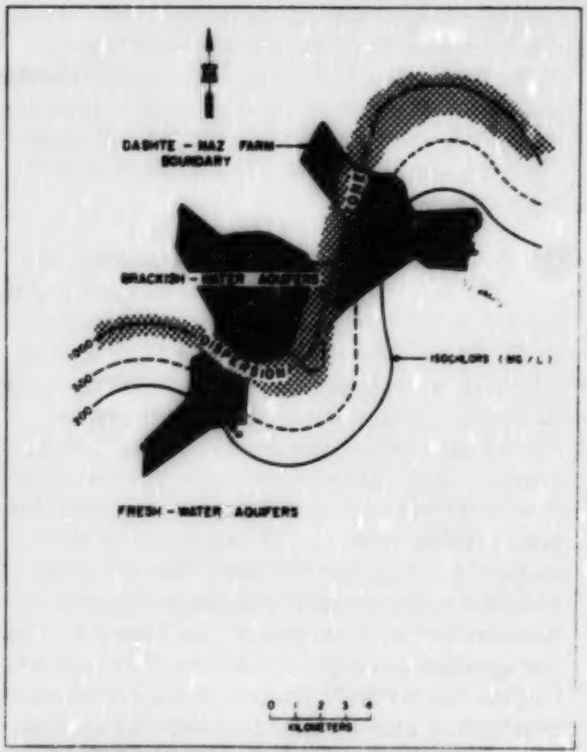

Fig. 6. Interface between fresh water and salinated aquifers in Dashte-Naz.

meters) consisting of the high-pressure connate-water sands.

Identification of the areal variation of salinity in the upper fresh-water zones was the major purpose of the exploration works. Defining the zone of dispersion between the fresh-water aquifers lying near the alluvial fans and the brackish-water zones existing in the low point between the fans took many months of careful drilling, testing and evaluation. Coring, vertical electrical conductivity profiles and electrical resistivity, gamma ray and gamma ray neutron logs were used extensively to locate this zone (see Figure 6).

Isochlor maps combined with aquifer test analyses showed that the most promising area for construction of a ground-water barrier and artificial recharge project would be in the southwest portion of the farm. Although the northeast portion of the farm showed a similar fresh-saline water dispersion zone, it was discarded as the first choice for an artificial recharge and recovery scheme due to the fact that the aquifers were less distinct and significantly lower yielding.

MANAGEMENT AND PROTECTION OF THE GROUND-WATER RESOURCES IN THE DASHTE-NAZ AREA

The Pumped-Storage Concept as Applied to Ground-Water Reservoirs

The principle of pumped-storage hydroelectric plants is common practice in many surface-water reservoirs. These plants maintain enough storage during off-peak hours for use during peak hours of the same day. The pumped-storage plant generates high-value peak power from the headwater to the tailwater pool when the demand is high. Low-value power is then used to pump water from the tailwater pool to the headwater pool during periods of low demand. The value of the pumped-storage plant lies in its efficiency to convert low-value off-peak power to high-value peak power.

The Dashte-Naz ground-water barrier and artificial recharge project operates on a plan similar to the pumped-storage system of a hydroelectric plant. During the non-irrigation season (off-peak period) when the demand for water is low, water from outlying well fields is piped into the project area and stored underground for later use when water demand is high. The water is physically stored through a series of injection wells tapping the upper three fresh-water aquifers. The artificial recharge cycle lasts about nine months of the year (September through May). During the three-month irrigation season (June, July, August), the imported water is then used to help meet the field irrigation-water requirements and the previously artificially recharged water, recovered through extraction wells makes up the balance of the irrigation demand.

Field Irrigation-Water Requirements

As the first real solution to the problem of meeting the field irrigation-water requirements at Dashte-Naz, supply wells were drilled in the vicinity of the Tajan and Neka Rivers. These wells were out of the geohydrologic influence of the farm area due to high natural recharge near the rivers and thus posed no danger of inducing saline-water encroachment. The pumped ground water from these outside source wells is conveyed into the farm area irrigation canals through a system of pipelines.

The water presently imported is able to provide the necessary irrigation water for most of the farm area. The eastern portion of the farm is supplied by imported ground water pumped from ten wells drilled near the Neka River. The western farm area is irrigated by ten wells drilled near the Tajan River and conveyed to the farm by two pipelines. The central portion of the farm has a deficiency in water supply which will be met by utilization of the ground-water resources available within the farm boundaries itself.

Injection Well Barrier

To meet the water requirements in the central farm area, the concept of a ground-water barrier was conceived. The design provides irrigation water by storing imported water, as obtained from the outlying well fields, underground into the three fresh-water aquifers of the southwest area and pumping it out later when needed for irrigation. A total of five production wells are needed in the southwest area to recover the artificially recharged water. These recovery wells will then export the water to the central farm area through a pipeline and pumping station (see Figure 7). In order to protect the southwest well field from the threat of saline-water encroachment during the pumping season, a ground-water barrier of approximately 2,000 m in length has been constructed. The barrier consists of nine injection wells spaced approximately 250 meters apart. The alignment of the barrier, as seen on Figure 4, parallels the isochlor lines and is normal to the direction of encroachment.

The spacing, injection rates and total length of the barrier were designed based on results

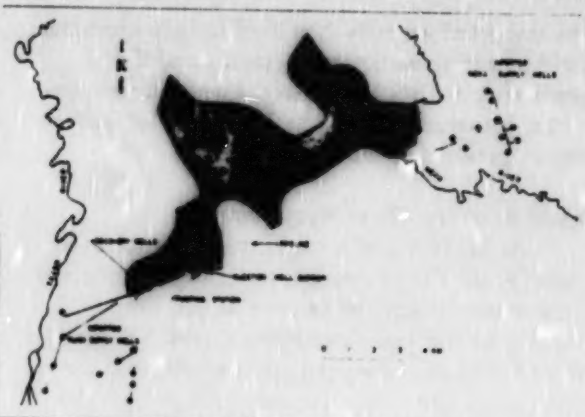

Fig. 7. Dashte-Naz area showing injection wells, recovery wells and imported water supplies.

from interference testing and mathematical simulation models of the area.

The injection wells are constructed so that selective injection can be made into individual or various combinations of the three aquifer zones in the event that salination occurs in any one or all of these zones. The three aquifers are cemented off from each other and pneumatic packers installed to prevent cross flow. At the present time, intermixing of the aquifer zones is permitted through the screen sections as the ground-water quality in all three zones is excellent. Figure 8 shows a typical cross section of an injection well.

Source of Water for Irrigation and Artificial Recharge

During the early planning stage, several sources of water for artificial recharge were investigated. Surface-water diversions were discarded due to irregular flows and sedimentation problems. Ground water imported from areas outside of the influence of the Dashte-Naz farm area proved to be the most practical and reliable source of injection water.

Three main well fields were thus developed and connected to the Dashte-Naz farm area through a pipeline conveyance system. The ten wells developed near the Neka River on the eastern side of the farm, provide an instantaneous flow of 2,400 m³/hr.

Two separate pipelines from the western side import water from ten wells tapping aquifers near the Tajan River, producing 2,500 m³/hr.

The Artificial Recharge and Recovery Scheme at Dashte-Naz

Operation of the ground-water barrier consists of injecting water imported from the Tajan River

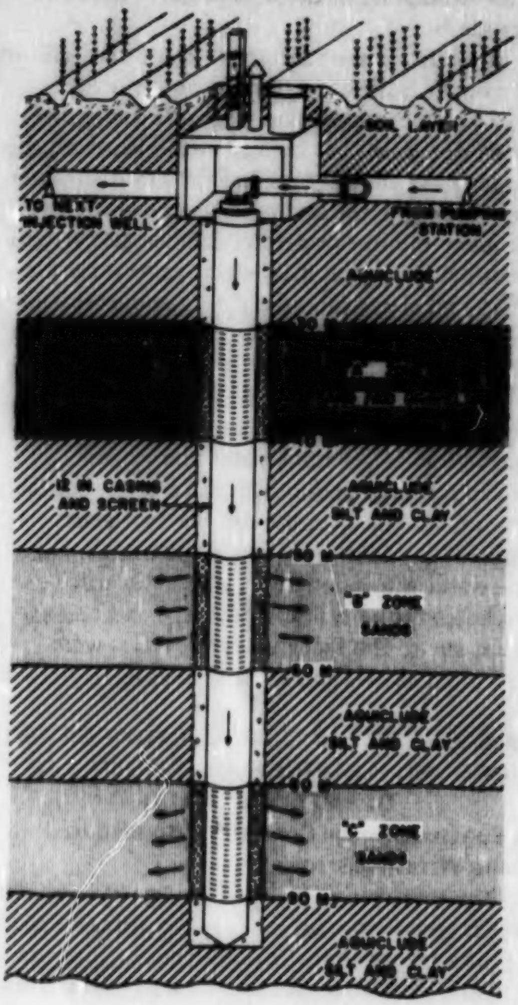

Fig. 8. Cross section of typical injection well.

supply wells during the non-irrigation season (September through May). At the end of this 9-month injection cycle, a rise of approximately 8 meters will have occurred in the piezometric surface in the vicinity of the southwest barrier wells. At this time, the water previously piped from the Tajan River wells will discharge into the irrigation canal system to meet the field irrigation-water requirements of the southwest area.

Also at this time, the five production wells located in the southwest farm area will be turned on to recover the previously artificially recharged water. The collective discharge from the recovery wells will be piped into the central farm area to meet the irrigation-water requirements there. The ten supply wells near the Neka River will also be

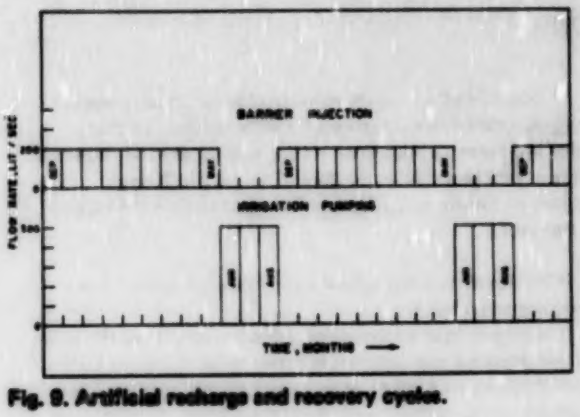

Fig. 9. Artificial recharge and recovery cycles.

started and their water used to meet the field irrigation-water requirements in the eastern farm area (see Figure 9).

Supervisory Control System

To properly regulate, control and monitor the complex system of injection and production wells throughout the year, a Supervisory Control System is presently being installed whereby pump motors, pipeline valves, and water-quality sensing devices can be interrogated and remotely controlled through a central computer housed in a new farm headquarters building. From this central control room, continuous records will be kept on all well flows, pressures and water-quality changes, so that immediate regulation can be made should adverse conditions arise.

The Supervisory Control System basically consists of five remote stations which relay digital information to a Supervisory Control Master Unit which in turn conveys the information to the computer (see Figure 10). The computer converts these digital data into pressure readings, valve set points, flow measurement and electrical conductivity and chloride ion concentrations and stores the data on memory disks generating a historical record of operation.

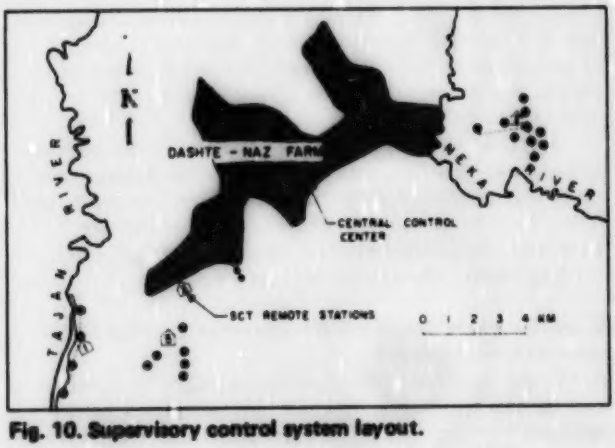

Fig. 10. Supervisory control system layout.

The real-time data are continuously checked against minimum and maximum allowable ranges, and alarm messages printed on the operator console if the ranges are exceeded. Background programs consisting of ground-water simulation models and pipeline hydraulic flow models are automatically loaded into the computer if conditions warrant new operating set points.

The operator of the computer in the central control center can, at any time, selectively interrogate remote stations to obtain information on any specific measurand. Predictive models can also be run simultaneously during real-time data acquisition in order to anticipate operating regimen changes due to variations in injection well pressures, flow rates, or pipeline flow variations.

The Supervisory Control System at Dashte-Naz represents application of modern-day-computer technology to the field of water-supply management and allows a degree of flexibility and accuracy of operation virtually unattainable by conventional water-management techniques.

SUMMARY

Protection of fresh-water supplies from saline-water encroachment has been practiced for many years throughout the world. However, the results from the Dashte-Naz project will be the first such in Iran to illustrate conservation and management of ground-water reservoirs in salinated areas. Results from the Dashte-Naz ground-water barrier and recharge project will be used to design ground-water barriers in other Caspian Sea coastal areas and along the Persian Gulf. Operation of the project requires extensive and careful engineering and management if the project is to become a smooth and efficient operation. One of the most important tools to ensure this efficiency is the remote control monitoring of all key elements in the project. A Supervisory Control System which monitors and regulates the artificial recharge and recovery scheme is therefore indispensable. In addition to regulating water flows, hydrometeorological and soil moisture information will also be relayed to the Supervisory Control System computer to aid decision-making during times of irrigation and harvesting.

In summary, the Dashte-Naz ground-water barrier and recharge project is a practical example of applying new methods and technology to the field of ground-water resources management.

DISCUSSION

The following questions were answered by Dennis E. Williams after delivering his talk entitled "The Dashte-Naz Ground-Water Barrier and Recharge Project."

Q. by K. Childs. *What was the total cost of the project, and who paid? What percent of injected water was recovered?*
A. The total cost of the project including the exploration, final design, and supervision of construction was approximately $6,000,000. The Iranian Government funded the project and the Ministry of Energy administered the project.

The percent of injected water recovered will not be accurately known until after several years of Project operation but from the initial tests, it is predicted that 85 to 90 percent of the injected water will be recovered by inland well fields.

Q. by Bill Wilmitts. *Was a barrier well line also considered for the east well field?*
A. Yes, an injection well barrier was considered to protect the northeastern portion of the Dashte-Naz ground-water reservoir; however, as the first stage it was felt the southwest reservoirs were more geologically suited for ground-water storage due to better definition of the aquifer systems. In the future, an injection well barrier will most probably be constructed to protect the northeast aquifers as well.

Q. *Are the sediments you referred to called "the Kuwait group" as the oil companies named it?*
A. No, the sediments underlying the Dashte-Naz farm area are mainly interbedded Quaternary alluvial and marine deposits and, as far as I know, the N.I.O.C. (National Iranian Oil Company) has not given any name to these formations.

Q. by C. Roberts. *Is there a functional relationship between injection rate and assimilation rate of the fresh water into the salt-water wedge?*
A. The quantitative relationship between injection rate and loss of fresh water to the dispersion zone was determined analytically through the use of a distributed parameter digital computer simulation model and verified somewhat experimentally with field tests.

Q. *How many acres are mechanically irrigated today?*
A. Approximately 3,000 hectares or 7,400 acres are irrigated by a combination of moveable sprinklers, center-pivot sprinklers and conventional ditch and furrow irrigation.

Q. by R. P. Chagnon. *Did you have any problems with cross-flow between the three aquifers; are they hydrologically connected and how long will the well last?*
A. The three aquifer systems are hydraulically separated from each other by thick aquicludes of silt and clay. This has been verified by different piezometric surfaces measured in the individual aquifers. At present, cross-flow is taking place between the aquifer systems through the screen sections but this is not felt undesirable as all three aquifers presently contain fresh water and the pressure gradients are small. However, should one of the aquifers become salinated it will be mechanically isolated by a packer to prevent an undesirable cross-flow condition.

The life of an injection well depends mainly upon the clogging rate of the screen sections. It is foreseen that redevelopment at least once a year will be needed; however, some injection wells in the West Coast Basin Barrier Project of Southern California have not been redeveloped in ten years.

Q. *What about vertical upward intrusion underneath the pumping well field?*
A. It is hoped that upconing of saline waters from the underlying connate-water aquifers will not take place due to the careful attention paid to completion of the injection and recovery wells. A thick aquiclude separating the fresh and connate zone exists between 100 meters to approximately 130 meters depth and unless this aquiclude is penetrated no upconing of undesirable waters should take place.

Q. by J. Brown. *Is there any danger of areal subsidence caused by the release of artesian pressure of the connate water?*
A. Since there will be no pumping of the connate-water aquifers there will be no release of artesian pressure in these zones. As previously mentioned, only the upper 100 meters of fresh-water aquifers will be used for the artificial recharge and recovery scheme.

Q. by Don Runnells. *Do the native people have the training to maintain this complex system or is it the plan to keep on consultants indefinitely?*
A. All Iranian government contracts call for training of local personnel to properly operate and maintain the project once the foreign consultants terminate their work. For the Dashte-Naz Project the consultants have foreseen training at least eight to ten qualified Iranian personnel to operate the system. Normally, a lesser number of people would be trained for a project of this size but, due to the sophistication and complexity of the Dashte-Naz Project, it is felt advisable to train as many as possible.

Q. by Jean Schmidt. *By what mechanism does salt water migrate out of the clays into the sands at depths of 100 meters or so?*
A. During the accumulation of sediments in the region, water of deposition was trapped in the sediments due to this process. Progressively increasing pressures caused the by-then salty waters to be "squeezed out" of their clays and silts migrating into the adjoining sand layers which, owing to their properties, were compressed much less than the finer-grained silts and clays.

As this mechanism continued, the sands received more and more saline waters with internal pressures progressively increasing as the overburden became thicker. The end result is that these lower zones now contain fossil or connate water with the internal aquifer pressures quite high, often reaching several atmospheres in some areas.

Q. by Thomas R. Schultz. *Is this system economical with such costly technology?*
A. No, this system is not economical as judged by benefit-cost analysis techniques. However, this system was never intended to be economical as it is a research project on the

practical development of ground-water resources in the vicinity of salinated zones. The results from this project will be used to design other recharge and recovery schemes not only along the Caspian Sea coast but the Persian Gulf as well. In areas where this method of ground-water conservation and management is the only alternative left for water supply, the benefit of just having the water available cannot be judged by standard economic indicators.

Q. by W. B. Wilkinson. *What is the suspended solid content of the recharged water? Is there any danger of clogging of the wells and, if so, are back washing facilities available?*
A. The recharged water will be imported ground water and the suspended solid content (after initial start-up of the wells) will on the average be below 1.0 mg per liter. At least once a year, or when the injection pressures rise and injection rates decline, backwashing facilities will be accomplished by mechanical surging and airlift pumping. Chemical treatment will also be performed.

Q. by Dale Ralston. *What is the approximate water cost per acre irrigated?*
A. The specific construction cost would be approximately $810 per irrigated acre. The annual operating cost of the supply-well system is approximately $50,000. This amounts to approximately $7 per acre irrigated. However, these figures do not reflect the cost of building the existing irrigation and drainage canal and road system, but only reflect the civil and appurtenant works connected with the ground-water barrier and recharge portion of the project.

Injection/Extraction Well System—A Unique Seawater Intrusion Barrier[a]

by N. Thomas Sheahan[b]

ABSTRACT

A multiple-aquifer system in the bayfront area of Palo Alto, California is being intruded with seawater from San Francisco Bay. In order to combat this potential degradation of the ground-water supplies in the area, a sea-water intrusion barrier is being constructed consisting of a series of injection wells used to inject 2.0 million gallons per day (7.6 X 10^6 l/d) of reclaimed wastewater into a shallow aquifer. The injected water is subsequently removed by a similar system of extraction wells to avoid any possible degradation of the water-supply aquifers from this source, and to allow reuse of the reclaimed wastewater. The investigation phase included test drilling, aquifer testing and injection testing to determine the feasibility of the injection/extraction (I/E) concept. The number, spacing and location of I/E doublets were optimized using a digital computer model. The double-cased, double-screened wells were constructed using corrosion-resistant materials and were designed for ease of routine maintenance. In operation, injection and extraction will be computer controlled by sensing piezometric levels in a series of monitor wells. Water pumped from the extraction wells will be sold for industrial and agricultural purposes. The I/E well system has been approved for 87½ percent Federal and State grant funding.

INTRODUCTION

Among the objectives of the Santa Clara Valley Water District is the prevention of degradation of ground-water quality in the aquifers supplying various municipalities and industries in their district. The quality of ground water for these purposes is generally good. In a few locations, however, aquifers once used to produce good water have become intruded with salt water. One such area is the shallow aquifer, above 150 feet (45.7 m) depth, around South San Francisco Bay. The aquifers deeper than 150 feet (45.7 m) still produce good water and are apparently protected by thick, clay aquicludes which have prevented salt water from the bay or from shallower aquifers from migrating to these deeper producing zones. Heavy pumping in the deep aquifers has produced a hydraulic gradient in the area away from the bay toward the producing well zones. Continued heavy pumping from the deeper aquifers may, in time, produce a degradation of these aquifers due to salt-water migration from the shallower zone or from the bay. Recognizing the need for control measures to reduce the salt-water intrusion into the shallow aquifers, a system has been designed which will incorporate injection of water into the shallow aquifers by wells in order to control the seawater intrusion.

The proposed salinity intrusion barrier contemplates the use of a series of injection wells and extraction wells located in pairs, termed doublets. These doublets form two lines of injection and extraction wells located essentially parallel to the bay front. This arrangement will allow injection of reclaimed water into the injection wells and the complete removal of this water through the extraction wells.

An investigation of ground-water conditions in the area was prefaced by a review of the physical environmental characteristics of the area in order to provide a basis for evaluation of changes in the ground-water regime.

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[a] Presented at The Third National Ground Water Quality Symposium, September 15-17, 1976, Las Vegas, Nevada.

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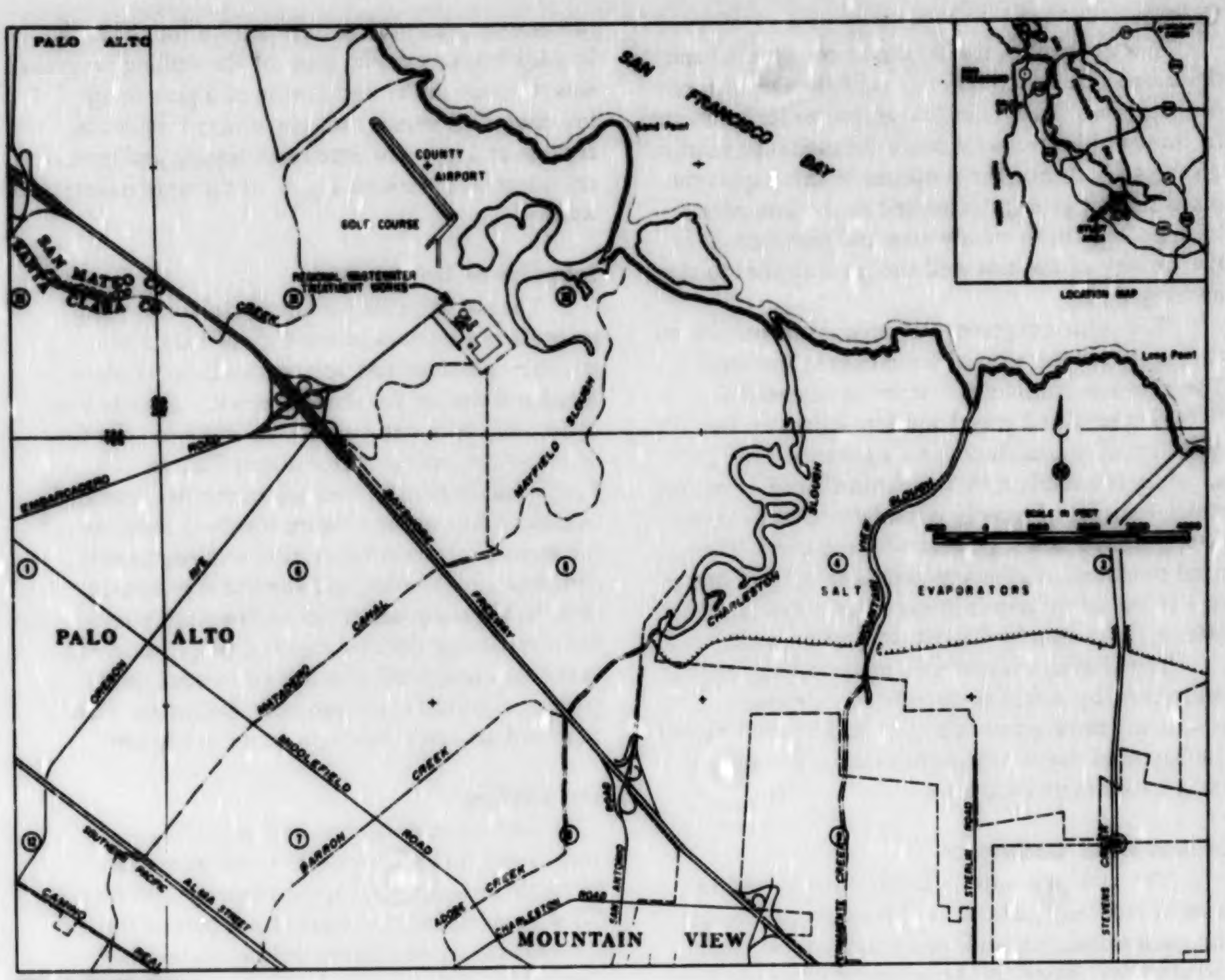


Fig. 1. Study area.

Study Area

The study area is located in the northwesterly corner of Santa Clara County, in the City of Palo Alto, along the south shore of San Francisco Bay. As shown in Figure 1, the area is bounded approximately by San Francisco Bay on the east, El Camino Real on the west, the City of Mountain View on the south and the City of Menlo Park on the north.

Topography

The study area is composed principally of flat lowlands and marshy baylands adjacent to San Francisco Bay. Generally the topography slopes northeasterly towards San Francisco Bay, and elevations in the area range from mean sea level at the bay front to a maximum of approximately 25 feet (7.6 m) above mean sea level in the western portion of the study area. The portion of the study area between Bayshore Freeway and the San Francisco Bay coast has been largely set aside by the City of Palo Alto for Baylands Park Preserve. There are, however, several areas of commercial and professional development near Embarcadero Road and along the Freeway.

Climate

The area exhibits normal ocean-moderated conditions. Temperatures in the area range from a mean of 47° F (8.30° C) in January to 66° F (18.90° C) in July with a mean annual temperature of 57° F (13.90° C). Average annual precipitation in the study area is approximately 15 inches (38.1 cm) although in the higher ground to the west of the study area, the average annual precipitation amounts to as much as 40 inches (101.6 cm). The higher rainfall to the west of the study area accounts for the extensive drainage patterns present and is the source of recharge water for the ground-water aquifers in the area.

Geology

The geology of the area has been studied and described extensively in reports of the Department of Water Resources (1967) and the reader is referred to these publications for more detailed information on the subject. For the purposes of this paper, the discussion of geology is limited to the immediate local geology in the study area and particularly in the vicinity of the test well site, as it applies to this investigation.

The principal geological deposits of interest to this study are the alluvial sediments in the area. The alluvium consists of a series of essentially flat-lying sand and gravel aquifers separated by extensive clay aquicludes and aquitards. The alluvium is underlain by the Santa Clara Formation which consists of loosely consolidated to compact, poorly-sorted sands, gravels, silts and clays. The total thickness of alluvium and Santa Clara Formation in the study area is in excess of 1,000 feet (305 m) as shown by logs of wells in the vicinity.

The alluvium is overlain, in portions of the study area, by soft, low-density clay deposits commonly referred to as bay mud. The thickness of the bay mud deposits is quite variable and ranges from 0 to 20 feet (6.1 m).

Ground-Water Occurrence

The principal waterbearing deposits, and the ones of most significance to this investigation, are the unconsolidated sands and gravels occurring in the alluvium. Based on the grouping of aquifers within the alluvium, the south bay area has been divided into various subareas which group areas of similar waterbearing formations (DWR, 1967). The study area includes portions of the Niles subarea and the San Francisquito subareas. Although the alluvium extends to approximately 1,000 feet (305 m) depth in the area, the aquifers of particular importance to this study occur in the upper 200 feet (61 m) of these deposits. Salt-water intrusion has occurred in the shallow deposits but the deeper deposits in the study area appear to contain fresh water. For this reason, only the aquifers occurring between ground surface and approximately 200 feet (61 m) depth are considered in detail here.

FIELD INVESTIGATION

In order to produce detailed information concerning the ground-water conditions and geology in the area of proposed injection of water for the seawater barrier, a program of test drilling and aquifer test analysis was developed. The site for initial test drilling was selected in order to provide the most representative information for the area. In addition, one of the goals of the drilling program was to result in the installation of a permanent injection well which could be utilized for initial testing, and later for continued testing and monitoring when incorporated as one of the operational wells.

Selection of Drilling Site

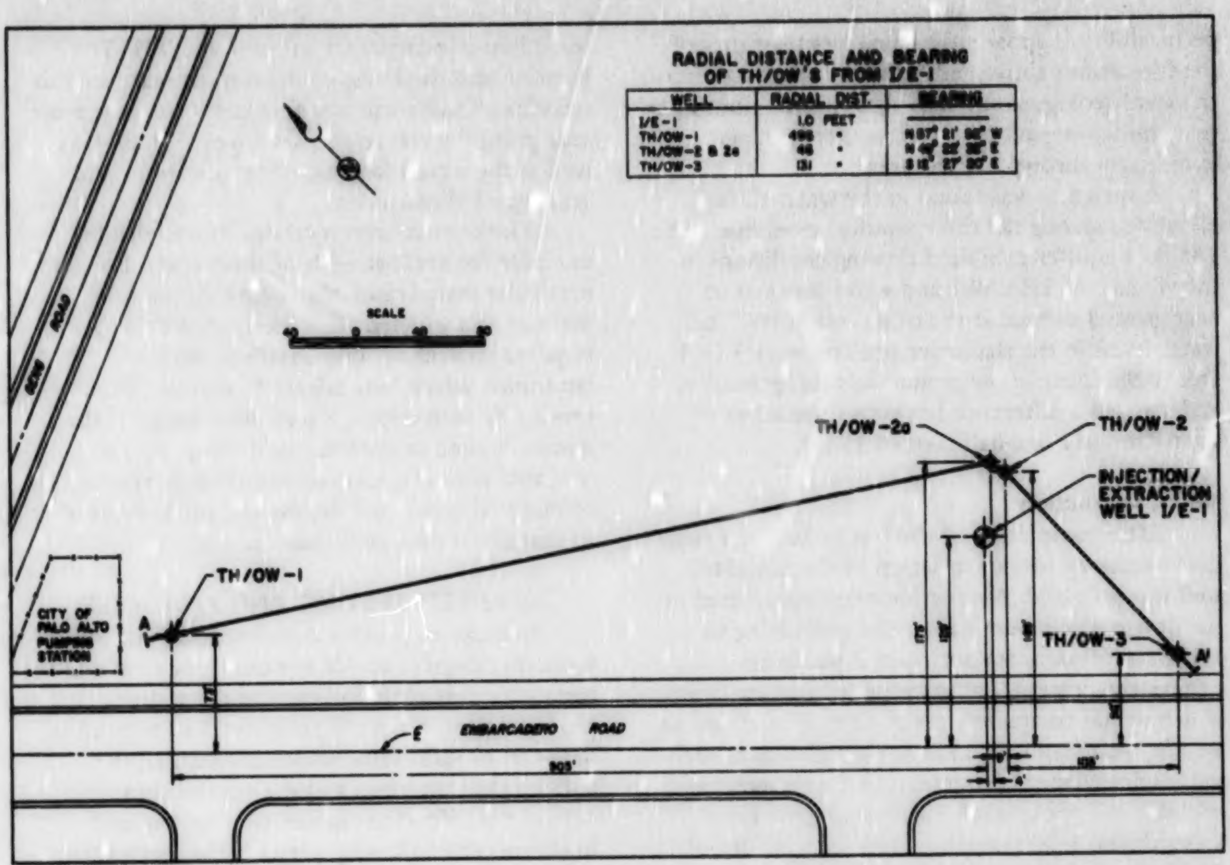
The site selected for the initial test drilling program is a portion of the City of Palo Alto property north of and adjacent to Embarcadero Road and east of Bayshore Freeway. The site was convenient to a water-supply source from the City of Palo Alto and for water disposal along Embarcadero Road. In addition, the site provided sufficient area in the vicinity for the installation of the second injection/extraction well required to complete the doublet, and the site is reasonably close to the proposed water reclamation plant. Examination of drillers' logs and geophysical logs in the area indicated that this site would show geological conditions reasonably similar to those expected generally throughout the study area.

Test Drilling

The test drilling consisted of drilling mud-rotary holes, 8-inch (20.3-cm) minimum diameter, to a depth of approximately 200 feet (61 m) and taking undisturbed samples of the soil at various locations during drilling. In test hole-observation well number 1 (TH/OW-1), undisturbed samples were taken approximately every 5 feet (1.5 m) in order to obtain the maximum amount of information concerning the subsurface. Samples of the materials encountered during drilling were examined on site both visually and with a mechanical sieve apparatus to determine grain-size distributions, material types, consistency and other soil characteristics. After each hole was drilled, geophysical logging was performed in the well to verify the written descriptive log and to obtain additional information concerning material types and water qualities. Locations of test holes are shown on Figure 2.

Observation Well Construction

After each TH/OW had been drilled and geophysically logged, and an analysis made of the information obtained, various sand and gravel formations were selected for installation of piezometers for measuring water-level changes and obtaining water-quality samples. Piezometers were constructed of 1¼-inch (3.175-cm) polyvinyl-



RADIAL DISTANCE AND BEARING OF TH/OW'S FROM I/E-I

WELL	RADIAL DIST.	BEARING
I/E-I	I.O FEET	—
TH/OW-I	495 "	N 87° 21' 56" W
TH/OW-2 & 2a	46 "	N 49° 22' 35" E
TH/OW-3	131 "	S 18° 27' 30" E

Fig. 2. Test well site.

chloride (PVC) pipe with 1¼-inch (3.175-cm) PVC slotted well screens. The well screens were set and gravel-packed opposite to the particular aquifer formations to be monitored and these zones were sealed from other aquifer zones by cement seals.

A review of the geological information obtained during the test drilling and geophysical logging program indicated the presence of two shallow aquifers, herein referred to as the "20-foot" (6.2-m) aquifer and the "45-foot" (13.7-m) aquifer both of which are invaded by sea water, as evidenced by analyses of water samples. Below the 45-foot aquifer, there exists an extensive clay aquiclude to a depth of approximately 180 feet (54.9 m). Below the aquiclude, there occurs another producing aquifer, herein called the "185-foot" (56.4-m) aquifer. This aquifer contains fresh water, and analyses of water samples showed no evidence of seawater contamination. The locations of the various aquifers found in the test drilling and their lateral relationships are shown in the geologic section in Figure 3. Slight variations in the formations were observed between adjacent borings; for example at TH/OW-1, there appeared to be a merger of the 20-foot and 45-foot aquifers. It was not determined whether this merger was due to previous construction in the area or was a natural occurrence. Comparison of additional subsurface information available near the north edge of the site indicated that the 20-foot aquifer is underlain in all cases by a clay aquitard as shown on Figure 3. However, south and east of the site area, additional test borings previously performed for another project did not disclose the presence of the 20-foot zone. It was, therefore, inferred that the 20-foot zone is an interrupted lense, although the fact that it is intruded by seawater indicates that it is areally extensive in other directions.

Some variation in the 45-foot aquifer was demonstrated by the test drilling on site; however, the variation is essentially in aquifer thickness and not in material type. The aquifer is essentially a sand and gravel zone in all three test holes. The 45-foot zone, or lenses similar to this formation, occurs over a fairly widespread area.

The 20-foot aquifer and the 45-foot aquifer are separated, in all three test holes, by a zone composed of varying amounts of silt, clay and fine sand. This zone, which is termed "aquitard" on Figure 3, has a measurable permeability but the permeability is apparently much less than that of the formations above and below it. Examination of the areal geology confirmed that this relationship of two aquifers separated by an aquitard occurs extensively throughout the area.

A variation was found in the water-table elevations among the three aquifers examined. The 185-foot aquifer exhibited flowing conditions in the vicinity of TH/OW-1 and water levels at or near ground surface at the other test holes. The water levels in the shallower aquifers were 3 or 4 feet (0.9-1.2 m) below ground surface, generally, and showed a difference between themselves of approximately one-half foot (0.15 m).

Well Construction

After completion of the test holes, the design data necessary for construction of the initial I/E well was prepared. Among the items considered in the design phase were size of the well casing to allow ample room for anticipated pumping equipment, selection of materials of construction which would adequately resist normal corrosion as well as chemicals used in future development techniques, location, size and length of well screens and slot size openings to insure negligible head losses during injection and extraction, facility of future maintenance, and adequate sealing of aquifers.

Figure 4 shows the construction details of the initial injection/extraction well, I/E-1. The location and thickness of the cement seals used to separate the adjacent aquifers and to seal the zone near ground level are also shown on Figure 4, as well as the screen lengths, screen locations and gravel-pack dimensions.

There are various methods of well drilling available for shallow wells of this nature. For this particular installation, the reverse circulation method was selected. This method of drilling requires no drilling fluid additives, such as bentonite, which may adversely seal the water-producing formations. Excellent samples of the materials encountered during drilling are also available with this method which allow verification of material types, and depths and thicknesses of formations during well construction.

AQUIFER TESTING AND ANALYSES

In order to provide data necessary for the ultimate design of an I/E system, a series of aquifer tests were performed on each of the water-producing aquifers at the test site. The tests were designed to yield information concerning the aquifer characteristics and the relationship between aquifers in response to pumping and injection. The following presents the methods of testing, the analyses of the test data and a summary of the geohydrology of the site as determined by analyses of the aquifer-test data.

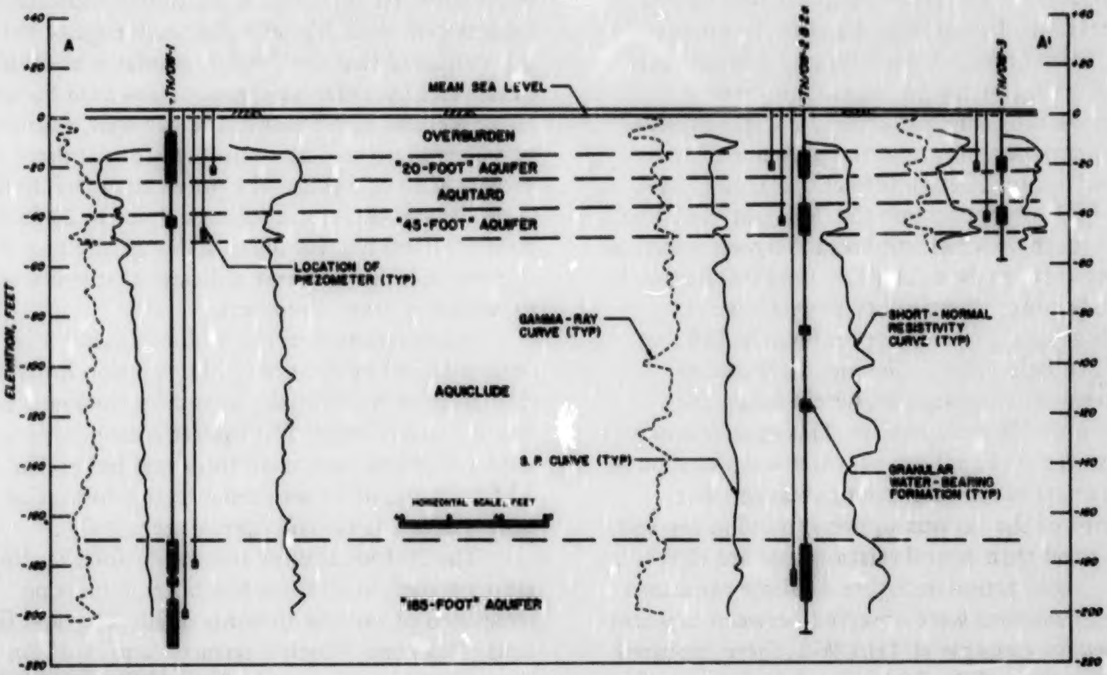


Fig. 3. Geologic section A-A'.

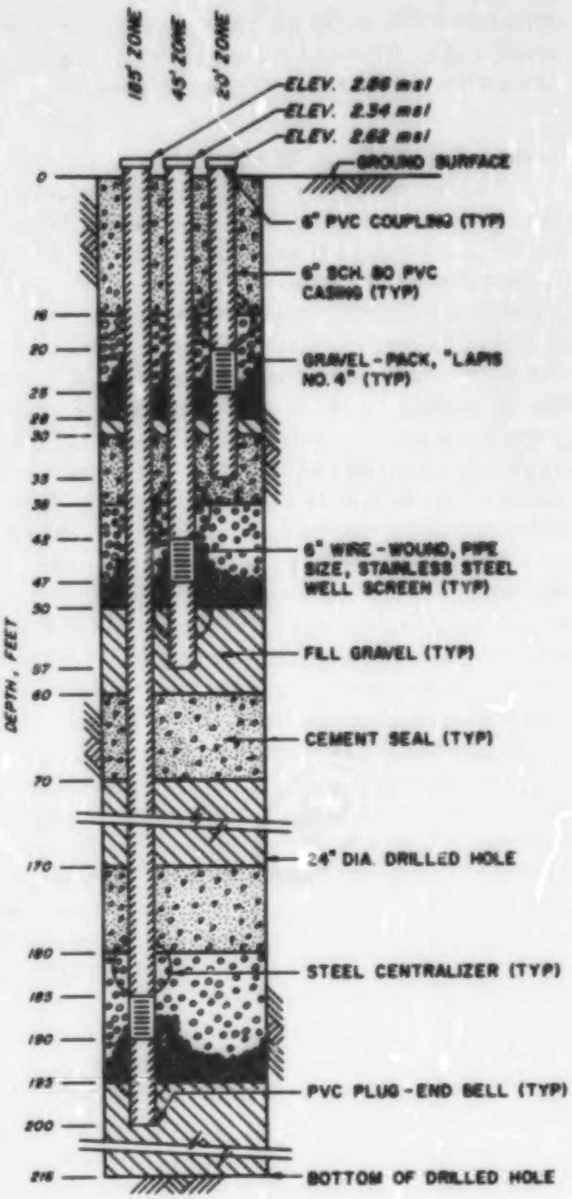


Fig. 4. Construction details, I/E-1.

Aquifer-Test Procedure

The aquifer testing of each screened aquifer was performed by pumping at a constant rate from I/E-1 for the period of approximately 24 hours, during which time changes in water level responding to the pumping were monitored in the pumping well, TH/OW's screened in that formation and in formations above and below the formation being tested. A series of water-level measurements were made in all the wells prior to starting the pumping portion of the test in order to establish a trend of water levels, and after pumping stopped, water-level monitoring was continued in all the wells to obtain data on the recovery of water levels after pumping.

In addition to the test wells and the pumping well at the test well site, water levels were measured in seven additional existing wells at various distances and directions from the test site in order to determine whether or not wells in these vicinities would respond to pumping at the test site.

Aquifer Test, 185-foot Zone

During the aquifer test of the 185-foot zone, the pumping rate in I/E-1 was held to a constant flow of approximately 165 gallons per minute (gpm) (10.4 l/s). Water-level changes were monitored in the pumping well, and TH/OW-2. In addition, water levels were monitored in the shallow aquifers in all three test holes and the I/E wells in the shallow aquifers. The water-level data were reduced to values of drawdown, corrected drawdown and recovery plotted against time since pumping began or time since pumping ceased. The drawdown values were corrected to account for projected trends of water levels prior to the beginning of the aquifer test. Pumping was continued for 24 hours, after which recovery was monitored for 24 hours. No response was noted in the shallow aquifers to pumping in the 185-foot zone. The data in these wells therefore were not considered in the following analysis.

The initial examination of the test data from the 185-foot zone indicated that the aquifer is an artesian, anisotropic aquifer, bounded on at least one side by a nearly impermeable boundary. The lack of response in the shallow aquifers during testing of this aquifer confirms the fact that it is a confined, artesian aquifer with no hydraulic communication to overlying zones.

Due to the confined nature of the aquifer in the 185-foot zone, the Theis nonequilibrium method of analysis was applied to the data (1935). The nonequilibrium formula is shown in Equations (1), (2) and (3).

$$s = 114.6\, Q\, W(u)/T \tag{1}$$

$$W(u) = \int_u^{\infty} (e^{-u}/u)\, du \tag{2}$$

$$u = 2693\, r^2 S/Tt \tag{3}$$

s = drawdown, in feet;

Q = discharge of well, in gpm;

T = transmissivity, in gpd/ft;

r = radial distance from pumping well, in feet;

S = storativity, a decimal fraction;

t = time since pumping started, in minutes.

In applying this method of analysis to the time-drawdown data for each observation well and the pumping well, it was found that the calculated coefficient of storativity increased with time. These variations have been shown to be characteristic of an aquifer which is bounded by one or more impermeable boundaries (Sheahan, 1967). In order to determine the distance to and direction of the apparent impermeable boundary, the theory of images (Ferris, *et al.*, 1962) was applied to the test data. This method, which is essentially a graphical superposition method, and a nongraphical method (Sheahan, 1967) of analysis for boundary conditions were utilized in determining the boundary characteristics of the aquifer.

The results of this portion of the analysis indicated that the 185-foot aquifer has an average transmissivity of 6,300 gallons per day per foot (gpd/ft) (78.12 m^2/d) and that there is apparently only one impermeable boundary reflected in the test data. The boundary was determined to be to the east of the injection/extraction well, a distance of approximately 900 feet (274 m).

In making calculations for the coefficient of storativity from the time-drawdown data for each of the wells, based on that portion of the data which did not reflect the boundary effects, a large variation in the magnitude of storativity was observed. This variation in apparent storativity is normally indicative of anisotropic conditions in the aquifer (Papadopulos, 1965). In order to calculate the directions of the major and minor axes of transmissivity and the values of the major and minor transmissivities in an anisotropic aquifer, a minimum of three observation points are required in addition to the pumping well. In this particular case, only two observation wells were available and therefore a complete analysis for anisotropy cannot be made. However, by assuming that TH/OW-1 was on a principal axis, a calculation of an approximate range of minimum-to-maximum transmissivity and a value for storativity can be made. Analysis under these assumptions indicated that the transmissivity may range from as low as 3,000 gpd/ft (37.2 m^2/d) to as high as 12,500 gpd/ft (155 m^2/d) and that the storativity is approximately 0.002 under these conditions. Based on the nature of the materials comprising this aquifer as shown on the drilling logs, together with the apparent thickness of the zone penetrated, the range of values of transmissivity, the average transmissivity and the apparent storativity values appear reasonable. It is noted that, if a value of storativity S, is assumed, calculation can be made of the other variables.

Aquifer-Test Analysis, 45-foot Zone

The test procedure employed in performing the aquifer test of the 45-foot zone was similar to that of the deeper, 185-foot zone, in that it consisted of pumping the 45-foot aquifer of the I/E well at a constant rate of 80 gpm (5 l/s) for a period of 24 hours, preceded by a rest period. During the rest period, the 24-hour pumping period and the 24-hour recovery period following pumping, water levels in the observation wells and pumping wells were observed and recorded. Water levels were monitored in both the 20-foot zone and the 45-foot zone during this test; however, the 185-foot zone was not continuously monitored since the aquifer test of that zone indicated it to be a confined aquifer and not hydraulically connected with the shallower aquifers. The water-level data obtained during the aquifer testing was reduced to values of drawdown and recovery which were plotted against the time since pumping began and time since pumping ceased, respectively. A graph of the aquifer-test data on a full logarithmic graph is shown in Figure 5. Due to the fact that the recovery readings were taken for a full 24 hours, corrections were necessary to these data to account for projected trends of drawdown due to the pumping portion of the test. These corrected recovery readings are also shown on Figure 5.

During the pumping and recovery portions of the test of the 45-foot aquifer, the water levels in the 20-foot aquifer were observed to respond to pumping in the 45-foot zone. The response of the 20-foot aquifer indicated conclusively that the 45-foot zone was obtaining recharge during the test from the shallower, 20-foot zone, and thereby must be considered a leaky aquifer, or more explicitly, a semiconfined artesian aquifer. Examination of the test data also indicated that the aquifer had not reached steady-state conditions and therefore must be considered in a nonequilibrium state for analysis.

In order to take into consideration the leaky artesian, nonequilibrium characteristics of the test data from this aquifer, the modified Hantush method of analysis (1955) was employed. The modified Hantush method utilizes a family of type-curves plotted according to Equations (4), (5) and (6).

$$s = 114.6\,Q\,H(u,\beta)/T \qquad (4)$$

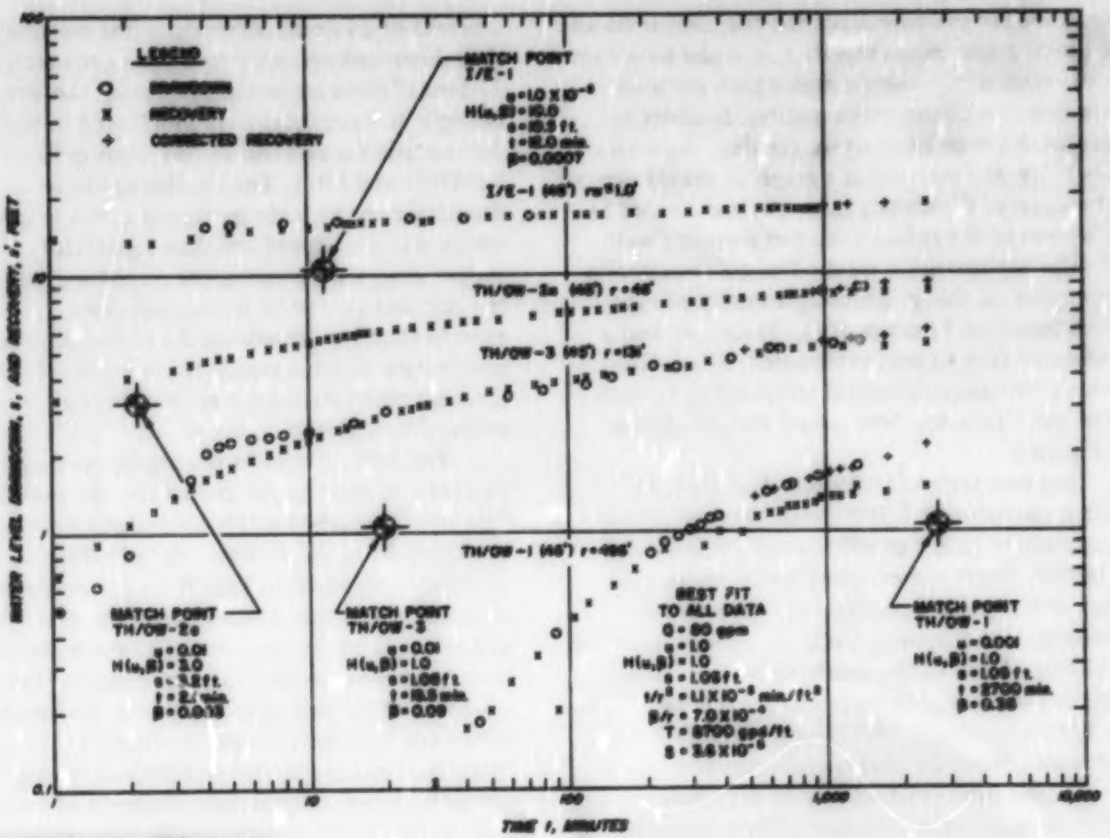


Fig. 5. Aquifer test, 45-foot zone.

$$H(u,\beta) = \int_u^{\infty} (e^{-y}/y)\ \mathrm{erfc}\ [\beta\sqrt{u}\ /\sqrt{y(y-u)}]\ dy \quad (5)$$

$$\beta = \frac{r}{4}\sqrt{\left(\frac{K'/b'}{T}\right)\left(\frac{S'}{S}\right)} + \frac{r}{4}\sqrt{\left(\frac{K''/b''}{T}\right)\left(\frac{S''}{S}\right)} \quad (6)$$

where

K', K'' = hydraulic conductivities of semipervious confining layers, in gpd/sq ft;

b', b'' = thickness of the semipervious confining beds, in feet;

S', S'' = storativities of the semipervious confining beds, as decimal fractions;

y = constant of integration;

and s, Q, T, r, S, and t are as previously defined.

The analysis of data by this method consists of attempting to match one of the family of type-curves to the data as shown on Figure 5 using a superposition procedure. When a position of best-fit is obtained, values of the parameters are read from the superimposed graph at a convenient match-point and the storativity, transmissivity, and leakance characteristics of the aquifer are calculated. In the particular case under analysis, it is obvious that there is leakage occurring only from above the 45-foot aquifer. This is substantiated by the fact that the 185-foot aquifer exhibited no hydraulic connection to the shallower aquifers, and the fact that there are no water-producing zones of any significant continuity occurring between the 45-foot aquifer and the 185-foot aquifer.

Concerning other physical limitations in this aquifer such as impermeable boundaries or anisotropy, the fact that excellent match was possible with the type-curves to each of the data plots indicates that there are no boundary conditions of any significance affecting the test data. An examination of the relationship between drawdown and distance from the pumping well shows that the drawdown varies essentially linearly with the logarithm of distance indicating that the aquifer is isotropic.

Although the data from each TH/OW may be analyzed individually to determine a unique value for each of the aquifer coefficients, it is of greater importance to evaluate all of the test data from all the observation points together, in order to obtain those values of the coefficients which are most representative of the entire aquifer. In order to perform this type of analysis, the data shown on Figure 5 were replotted as a graph of drawdown, s, and recovery, s', versus a value of time divided by the square of the radius from the pumping well, t/r^2. The superposition method of curve-matching was applied to this graph using a family of type-curves based on Equations (4), (5) and (6), and a position of best-fit was determined. From this analysis, the match-points corresponding to each set of test data were determined and are shown on Figure 5.

The composite aquifer-test data analysis yielded the value of 8,700 gpd/ft (108 m^2/d) for transmissivity, and a coefficient of storativity of 0.000036. These values appear to be reasonable based on the general geology and the relative geometry of the aquifer. Using an average thickness of 13 feet (4 m) for the overlying semipervious confining layer, as determined from the logs of the test holes and a value of β/r equal to 0.0007, as determined from the analysis, the storativity and vertical hydraulic conductivity of the semipervious confining layer were calculated. These calculations yielded a storativity of 0.001 and a value of vertical hydraulic conductivity of 0.032 gallons per day per square foot (gpd/sq ft) (0.013 m/d).

During the analysis of this set of test data, it was observed that the rate of change in water level in the 20-foot aquifer at I/E-1, TH/OW-1, and TH/OW-2a, with respect to the logarithm of time, were all essentially equal. The rate of change, or slope per log cycle time, of the data from TH/OW-3, however, was approximately twice the slope of the data plot from the other wells. Examination of borings made immediately south and east of TH/OW-3 indicate that the shallow, 20-foot aquifer, appears to be discontinuous in that direction. Therefore the increase in slope observed from TH/OW-3 may be a response to an impermeable boundary in the 20-foot aquifer. The leakance characteristics in this vicinity may also be different. This aspect of that aquifer is examined further in the analysis of test data from the 20-foot aquifer.

Aquifer-Test Analysis, 20-foot Zone

Due to the limited depth available for drawdown in the shallow aquifer, a very low constant pumping rate was used during this test. The I/E well in the 20-foot zone was pumped at an average pumping rate of 8.6 gpm (0.54 l/s) for a period of 24 hours after which the pump was shut down and recovery readings were taken for 17 hours following pumping. During the test, changes in water level were monitored in both the 20-foot aquifer and the 45-foot aquifer in all three TH/OW's and I/E-1. The small magnitude of drawdown in the wells produced a data graph which was considered less than satisfactory for proper analysis. However, the recovery data from the test was able to be monitored more exactly than the drawdown data and did produce usable data. Consequently, the water-level readings taken during the recovery period were reduced to a plot of recovery versus time.

The 20-foot aquifer essentially reached equilibrium prior to the end of the test period. The aquifer reached equilibrium due to the leakage occurring from the 45-foot aquifer below, and probably additional leakage from the overburden materials above the 20-foot zone. The short time period required for this aquifer to reach steady state is due partly to the low value of pumping rate used during the test, and also due to the apparent additional leakage from the overlying beds. Since the aquifer reached equilibrium during the test period, the leaky artesian formula of Hantush and Jacob (1955), which considers an aquifer approaching equilibrium was used in the data analysis. The leaky artesian formula is shown in Equations (7), (8) and (9).

$$s = (114.6\ Q/T)\ W(u, r/B) \qquad (7)$$

$$W(u, r/B) = \int_u^\infty (1/u) \exp(-u - r^2/4B^2u)\, du \qquad (8)$$

$$B = \sqrt{Tb'/K'} \qquad (9)$$

where all variables shown are as previously defined.

For this analysis, the data were replotted as recovery, s', versus a value of time divided by the square of the distance from the pumping well, t/r^2, and a family of type-curves, based on Equations (7), (8) and (9), was superimposed to a position of best-fit to the data. Utilizing the portion of the data prior to the point at which steady state was reached, values of transmissivity and storativity were determined. The transmissivity was found to be 3,500 gpd/ft (43.4 m^2/d) and the storativity value determined was 0.002.

Since the leakage to the 20-foot aquifer apparently occurs from both above and below that aquifer, and since the analysis method used assumes leakage from only one bed, it is not possible to determine exact values of the vertical hydraulic conductivity for the adjacent beds. However, the purpose of obtaining information concerning the aquifer characteristics is to allow prediction of future response to pumping or injection stresses on these aquifers, and a more usable parameter for this purpose is the range in values of B. From type-curve analysis, the value of B was determined to be in the range of 33 to 131 and averaged about 105.

Since, as indicated above, the latter part of the test data represents essentially steady-state conditions, it is possible to verify the value of transmissivity using a different method of analysis. The steady-state leaky artesian formula (Jacob, 1946) is shown in Equation (10).

$$s = (229\ Q/T)\ K_0\ (r/B) \qquad (10)$$

where

$K_0\ (r/B)$ = modified Bessel function of the second kind and zero order;

and s, Q, T, r and B are as previously defined.

The values of r/B obtained from the nonsteady-state analysis were plotted versus the steady-state, equilibrium recovery for each of the observation wells. To this data was matched a type-curve of the steady-state leaky artesian aquifer conditions. Analysis by this method yielded a value transmissivity of 3,650 gpd/ft (45.3 m^2/d) which compares favorably with the value of 3,500 gpd/ft (43.3 m^2/d) obtained from the nonsteady-state method of analysis.

Summary of Geohydrology of Test Site

The subsurface consists of essentially three separate aquifers within the depth of investigation. The deepest aquifer, the 185-foot aquifer, is a fully confined, artesian aquifer with an average transmissivity of 6,300 gpd/ft (78.1 m^2/d) and a storativity of 0.002, but is anisotropic in character and may have directional transmissivities ranging as low as 3,000 gpd/ft (37.2 m^2/d) to as high as 12,500 gpd/ft (155 m^2/d). The aquifer appears to be bounded by an impermeable hydraulic boundary located at a distance of approximately 900 feet (274 m) east of the I/E well. There was no hydraulic continuity observed between the 185-foot aquifer and the two shallower aquifers.

The shallow aquifers, consisting of a 20-foot aquifer and a 45-foot aquifer, exhibit a certain degree of hydraulic communication. Pumping of the 45-foot aquifer created a water-level response in the 20-foot aquifer. The 45-foot aquifer showed characteristics of a leaky artesian aquifer upon analysis. Although the pumping rate used in the 20-foot aquifer test was too small to allow any observable correlative response in the 45-foot aquifer, the fact that there is hydraulic communication between the two was established during the test of the 45-foot aquifer and verifies the apparent correlative response to pumping of the 20-foot aquifer.

Considering the fact that the depth to water level in both shallow aquifers is about the same, roughly 4 feet (1.2 m) below ground surface, it was anticipated that injection into the 45-foot aquifer at pressures greater than about 5 feet (1.5 m) of head, might eventually cause waterlogging of the surface of the ground due to leakage from the 45-foot aquifer, through the 20-foot aquifer, and up into the overburden material above this zone. The 20-foot aquifer, however, provides a means for controlling the waterlogging effect that might possibly occur. While injecting into the 45-foot aquifer at a fairly high rate, the 20-foot aquifer can be pumped at a much smaller rate, but a rate sufficient to control waterlogging in the area.

Injection Testing

After the completion of aquifer testing of the three aquifers encountered in the test well site, actual injection testing was performed on the 45-foot aquifer. The purposes for which the injection testing was performed were actually threefold. First, although the analysis of the aquifer-test data provided the information necessary to predict the results of injection, it is valuable to be able to confirm the accuracy of these predictive techniques. Second, injection testing produces additional, usable information not obtained during the aquifer testing which was of value in design and operation of the injection system. Third, and possibly most important, it is valuable to demonstrate that the proposed injection procedure will perform as expected under actual field conditions, and with complete control of the operation.

In designing the injection test, the first criteria of importance was a value of maximum injection pressure to be used during injection. In order to arrive at this maximum figure, the criterion was established that the injection pressure should be no greater than the net loading of soil at the point of injection. In order to calculate this, the static loading of soil down to the top of the aquifer was determined to be 35.3 pounds per square inch (psi) (2.48 kgs/cm^2). Due to the artesian nature of the 45-foot aquifer, the hydrostatic pressure in the formation is 15.6 psi (1.10 kgs/cm^2). The net static loading, therefore, is the difference between these two values, or approximately 20 psi (1.4 kgs/cm^2), at the design point. Based on the aquifer-test coefficients available, this value corresponds to an injection pumping rate of approximately 200 gpm (12.6 l/s).

In order to obtain additional data concerning the possibilities of waterlogging in the area, additional shallow borings were made in the vicinity of the test holes on site and were equipped with piezometers to a depth of approximately 3 to 5 feet (0.9 to 1.5 m). These piezometers were also monitored during the injection testing. In addition, concrete monuments were set in the soil at various locations around the test well site, and accurate elevations were determined at these monuments. The elevations of the monuments were monitored during the injection testing to determine whether or not heaving of the upper layer might be occurring due to the injection pressures.

Due to limitations in the availability of injection pressures at the site, the injection test was actually performed at an injection pressure of approximately 12 feet (3.66 m) and at a pumping rate between 90 and 100 gpm (5.7 and 6.3 l/s). The test was conducted using a constant-head, variable-capacity method which varied slightly from the aquifer test in that the aquifer tests were performed with a constant pumping rate and a variable head. The water used was fresh water from the City of Palo Alto municipal water supply, and was injected into the 45-foot aquifer at I/E-1. The test was continued for 48 hours under these conditions, and changes in water levels were monitored in the I/E well and the TH/OW wells in both the 20-foot and 45-foot aquifers. At the end of the 48-hour period, both the injection and the pumping operations ceased and recovery readings were obtained.

Examination of the test data at the end of 24 hours of injection showed that the water level in the 20-foot aquifer at TH/OW-3 had risen to a point approximately 6 inches below ground surface. By contrast, the water level in the 20-foot zone at TH/OW-2a, 85 feet (26 m) closer to the injection well, was still nearly 2 feet (0.61 m) below ground surface. This rapid rise at TH/OW-3, southeast of the injection well, correlated with what was expected in that area based on the aquifer-test analysis of the 45-foot aquifer which indicated the possibility of an impermeable barrier in the 20-foot aquifer to the southeast of the site. At that point in the test, the shallow, 3 to 5-foot (0.9 to 1.5-m) piezometers still did not show any waterlogging of the overburden materials.

After 24 hours of injection alone, and at the point at which the water level in the 20-foot aquifer in the vicinity of TH/OW-3 was 6 inches (15.2 cm) below ground surface, pumping was started in the I/E well at the 20-foot depth at a rate of approximately 8 gpm (0.5 l/s). This pumping continued for the next 24 hours, during which time no additional rise in the water level in the shallow aquifer at TH/OW-3 was observed. In fact, the water level actually receded approximately 0.15 feet (0.05 m) during that time. The shallow piezometers showed no response to injection during the entire 48 hours of injection, and there was no change in elevations of the concrete monuments observed during testing.

The injection test therefore demonstrated that fresh water can be injected into the 45-foot aquifer with no detrimental effects and that possibilities of waterlogging can be adequately controlled by pumping the shallow aquifer, thereby providing complete operational control of the injection system.

Prediction of Injection Response

As a result of the injection testing, the actual response of the 20-foot and 45-foot aquifers to injection of fresh water under the test conditions was known. The next step was to verify that this response could be accurately predicted based on the knowledge of the aquifer characteristics determined by the aquifer testing.

In order to calculate the expected response of injecting at a constant head, a constant pumping rate was selected based on the range of injection rates used during the injection test and a variable-head, constant-rate method, similar to that used in analyzing the aquifer-test data for the 45-foot aquifer, was employed, since under the test conditions at I/E-1, very little change in pumping rate was observed. The predicted response and the actual response of injection for two of the test holes in the 45-foot aquifer are shown on Figure 6. Similar data for the 20-foot aquifer for the same two test holes are shown plotted on Figure 7. As can be seen from the Figures, there is an excellent correlation of predicted response with actual response for the 45-foot aquifer, and for the nearest test hole in the 20-foot aquifer. The low magnitude of water-level rise in the 20-foot aquifer at TH/OW-1, located 496 feet (151 m) from the I/E well, may account for the slight deviation of

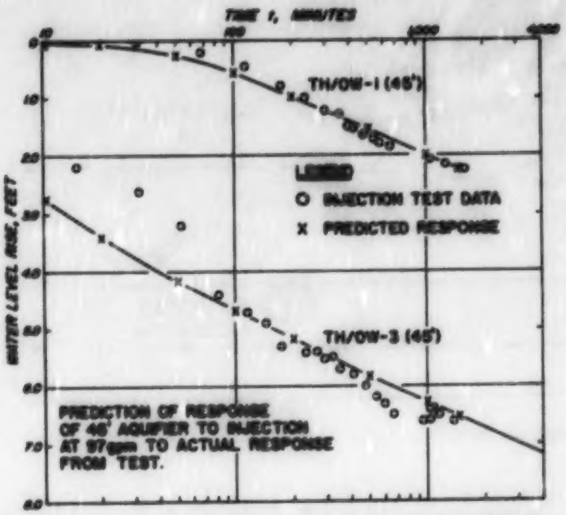


Fig. 6. Response of 45-foot zone to injection into 45-foot zone.

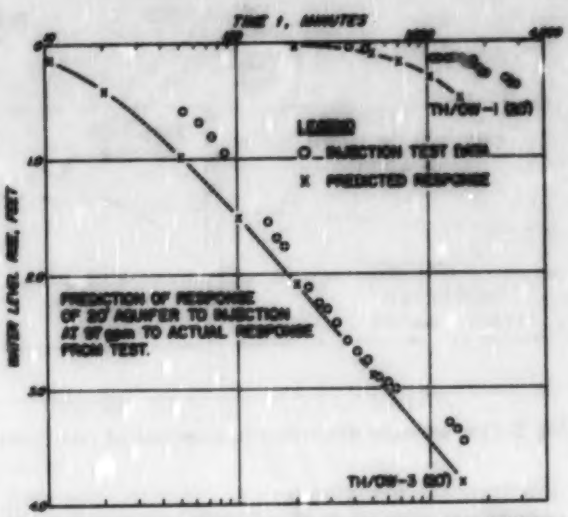


Fig. 7. Response of 20-foot zone to injection into 45-foot zone.

actual response to predicted response observed for that well. It is noted, however, that the predicted response is conservative in that it shows a greater rise of water level than actually occurred.

In summary, therefore, it can be said that the aquifer characteristics as determined by aquifer testing are suitable for accurate prediction of injection response both in the injecting aquifer and in the aquifer into which leakage is occurring. Even the effects of boundaries in the aquifer being recharged can be taken into account in making these predictions.

Water Quality

Throughout the course of the field investigation, water samples were obtained from the various monitoring wells for determination of chlorides and specific conductance. A summary of the water-quality data is shown graphically on Figure 8. As shown by the Figure, there is a general correlation between specific conductance of the water samples obtained and the amount of chlorides present in the samples. There is also an apparent grouping of samples which correlates with the location of the wells and the depth of the aquifers from which the samples were taken. The lowest chloride contents were found in the 185-foot aquifer at the test site. Chlorides in these samples averaged 28 milligrams per liter (mg/l). By contrast, the amount of chlorides in the shallow domestic wells, located approximately one-half to one mile farther away from the Bay than the test well site, showed an average chloride content of approximately 110 mg/l.

COMPUTER SIMULATION

The proposed operation of the injection system consists of injecting reclaimed water into a series of wells in order to hold back seawater intrusion, and to extract this water from a series of extraction wells in order to keep the reclaimed water from migrating out of the area of the injection wells and into the vicinity of producing water-supply wells. Given an injection/extraction doublet located in an aquifer whose characteristics are known, under a regional hydraulic gradient greater than zero, there is a limiting streamline, the path of which can be determined, which will

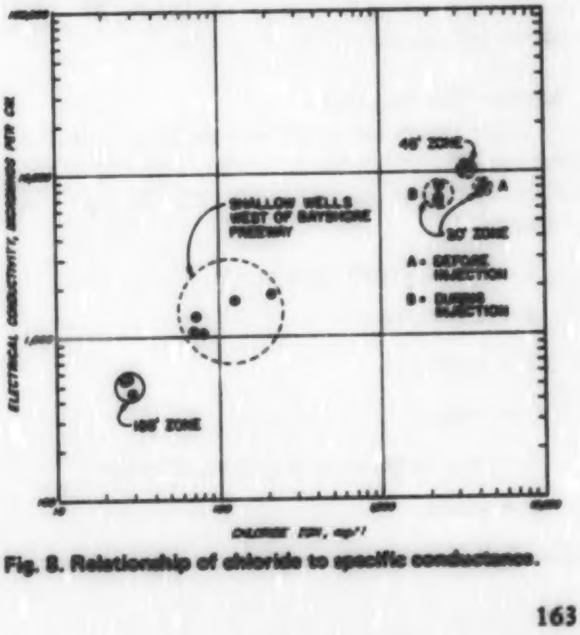


Fig. 8. Relationship of chloride to specific conductance.

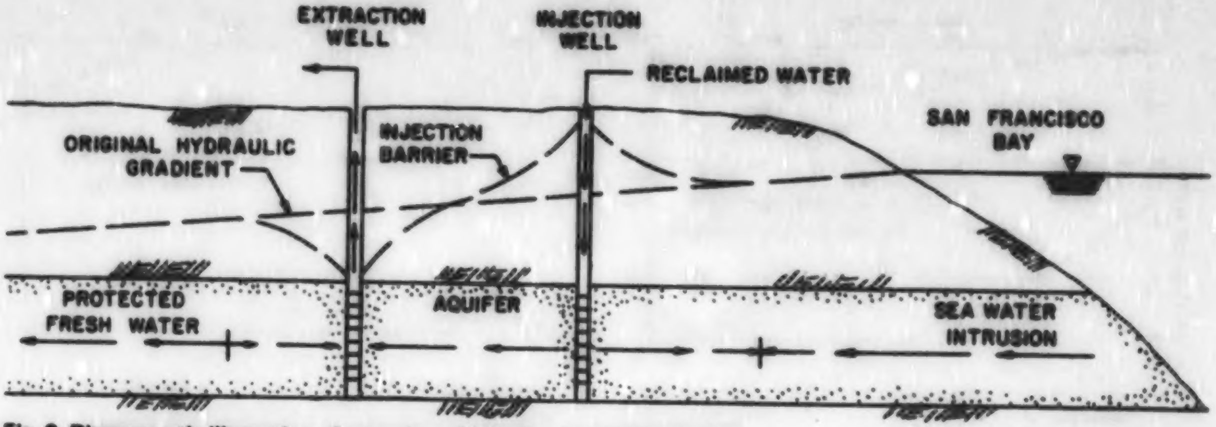


Fig. 9. Diagrammatic illustration of operation of injection/extraction doublet.

flow from the injection well to the extraction well, encompassing an area surrounding the doublet. Under the given circumstances, no particle of water can move outside of the perimeter of the area enscribed by the path of the limiting streamline. In addition, all other particles of water from the injection well will follow streamlines within that perimeter between the injection well and the extraction wells (D'Acosta and Bennett, 1960). A diagrammatic section illustrating this condition is shown in Figure 9.

Under simple, straightforward conditions, the path of the limiting streamline can be determined analytically. However, with conditions of multiple I/E doublets, complex aquifer conditions, and interaction between aquifers, the calculation of the path of the limiting streamline becomes a task more suitable to the use of a digital computer. For this reason, a computer simulation approach (Ramey, *et al.*, 1973) was used to determine the path of the limiting streamline under the conditions particularly applicable to the study area.

Aquifer Characteristics

The computer model was designed to include the aquifer characteristics from the 45-foot zone. These aquifer characteristics include the following values:

T = 8,700 gpd/ft (108 m^2/d);

S = 0.000036;

β/r = 0.0007;

Q = varies;

t = very large, to approach equilibrium;

r = varies.

The digital computer model essentially is a mathematical model applying Equations (4), (5) and (6), incorporating the above aquifer characteristics as determined from the aquifer testing.

In order to determine a value for the hydraulic gradient through the area which would be representative of the actual field conditions, the hydraulic gradients in various portions of the study area were calculated. These calculations were based on water-level elevations in various wells and the gradients varied from nearly zero to as much as 0.002 ft/ft (feet per foot).

Preliminary computer simulations indicated that the area encompassed by the limiting streamline was quite sensitive to variations in hydraulic gradients. It was determined that as the hydraulic gradient became greater, the area encompassed by the limiting streamline became smaller. Therefore, in order to present a conservative approach, the largest gradient measured was utilized in preparing the simulation model.

Injection/Extraction Doublet

The digital computer simulation model was first applied to the problem of determining the effects of spacing between the injection well and the extraction well, and variations in pumping rates at the wells. Since it was anticipated that the injection rate and extraction rate will be equal in actual operation, this relationship was also used in the simulation. Figure 10 shows the path of a particle of water along the limiting streamline in a single doublet oriented as shown with respect to the normal hydraulic gradient. In this instance, the limiting streamline traces the approximate shape of an ellipse. By applying the simulation model to various well spacings and pumping rates, the relative relationships among doublet spacing, pumping rate and area encompassed by the limiting streamline were determined. It was found that at a constant spacing, the area is proportional to the rate of pumping, such that if the pumping rate is doubled, the area encompassed by the limiting streamline is also essentially doubled. It was further determined that the area is roughly proportional to the product of the rate times the spacing. For example, the area encompassed by the limiting streamline for the case of injection and extraction wells spaced 1,000 feet (304.8 m) apart, pumping at 50 gpm (3.15 l/s), is the same as the area resulting from wells spaced 500 feet (152.4 m) apart pumping at 100 gpm (6.31 l/s). It was also shown, as mentioned previously, that the area is inversely proportional to the regional ground-water hydraulic gradient through the area.

The path of the limiting streamline becomes somewhat more complicated when multiple injection/extraction (I/E) doublets are considered. Obviously, if three doublets are spaced side-by-side, they will have the same effect on the limiting streamline as would a single doublet pumping at three times the pumping rate. On the other hand, if the doublets are spaced great distances apart, each doublet will operate as if the other doublets do not exist. However, in the intermediate zone between these two extremes, there is a particular spacing of doublets at which the path of the limiting streamline from the center doublet changes from a path completely encompassing all three doublets to a path that encompasses only the central doublet. This effect of doublet spacing on the path of the limiting streamline is illustrated in Figure 11. In Example A on this Figure, the doublets are relatively close and the path of the limiting streamline encompassing all three doublets follows essentially the same path as if the system consisted of only a single doublet pumping at three times the rate of each of the doublets shown. Example B shows three doublets spaced somewhat farther apart. The difference in shape of the area encompassed by the path of the limiting streamline in this example, as compared with Example A, shows the effects of the increased spacing. The effect of the spreading out of the doublets, or

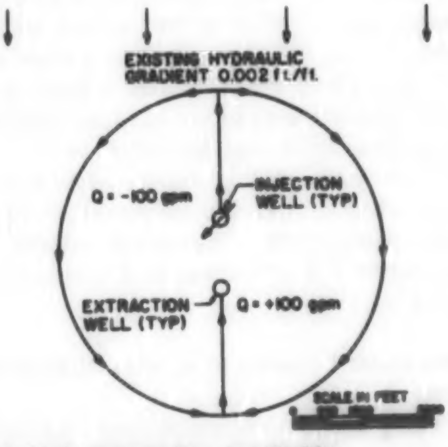


Fig. 10. Limiting streamline, one doublet.

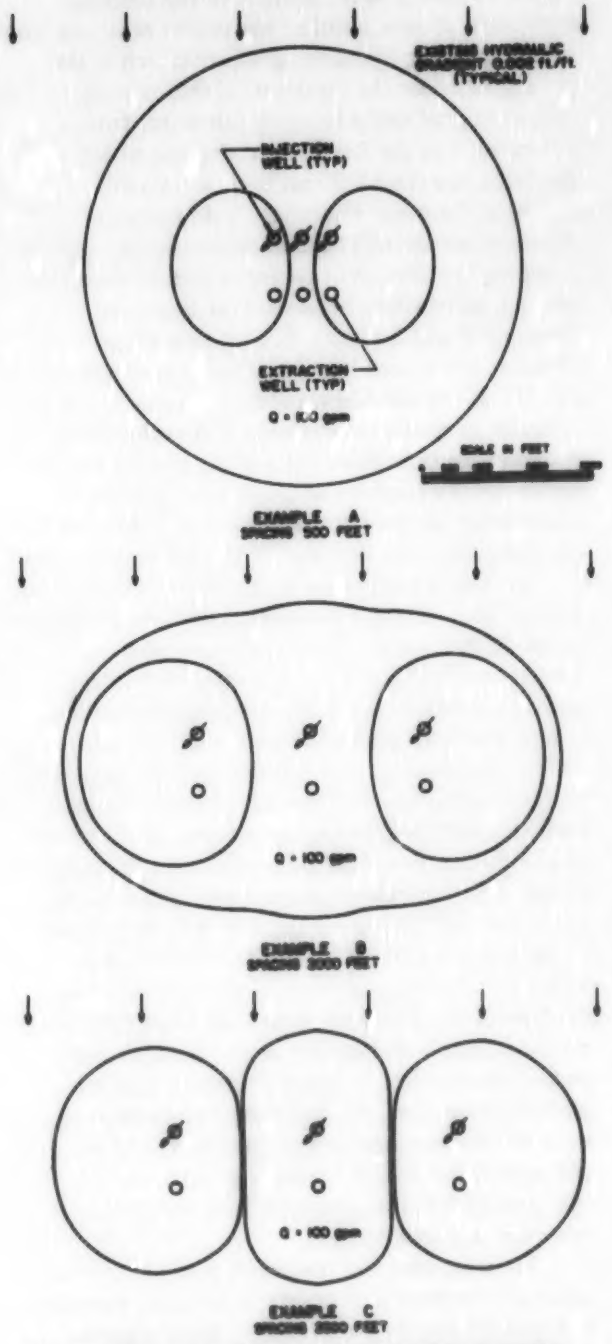


Fig. 11. Effect of doublet spacing on limiting streamline.

increase in side-to-side spacing, causes the limiting streamline path to approach the point at which it will change direction and pass between adjacent doublets. Example C shows the doublets located at a spacing just greater than the critical spacing at which the streamlines have, in fact, broken through or changed direction, and each streamline path is encompassing an area around only its own doublet.

It is noted that in the case of Example C in Figure 11, there is, in addition to the limiting streamline shown, another streamline resulting from the pre-existing hydraulic gradient which is also passing between the doublets. If this existing hydraulic gradient is bringing salt water from a source such as the Bay toward the line of I/E doublets, the doublets spaced as shown in Example C are not effectively holding back seawater intrusion. There is, obviously, for a given pumping rate and well spacing, a critical doublet spacing, somewhere between that shown in Example B and Example C, such that if the doublets are spaced less than that critical spacing, the line of I/E doublets will do an effective job as a barrier to seawater intrusion. It was therefore necessary to determine the critical spacing requirements for the location of I/E doublets, and the relationship of the critical spacing to either the I/E pumping rates, the number of I/E doublets, or both.

As stated above, the area encompassed by the path of the limiting streamline is directly proportional to the pumping rate and to the spacing between wells. Therefore, it is certain that the critical spacing between the doublets, depending as it does on the path of the streamline, is also dependent upon the spacing between the injection and the extraction well in each doublet. From an evaluation of the physical constraints of the study area and the geohydrologic parameters, the range of separation between the injection well and the extraction well in a given doublet was determined to be between 500 and 1,500 feet (152.4 and 457.2 m). For the purposes of this analysis, this dimension was assumed to be 1,000 feet (304.8 m). In addition, the aquifer characteristics and maximum injection pressure indicated that the pumping rates for both injection and extraction wells are in the range of 100 gpm (6.31 l/s) to 200 gpm (12.6 l/s). The pumping rate value of 150 gpm (9.5 l/s) was assumed as a reasonable value for this analysis.

To determine the relationship of the critical spacing between I/E doublets to both the pumping rate and the number of doublets, the computer simulation technique was first applied to a three-doublet system to calculate the critical doublet spacing for various pumping rates and various well spacings within each doublet. The results of this analysis indicated that, for a particular well spacing in a three-doublet system, as the pumping rate in both the injection and the extraction well increased, the critical spacing between doublets approached a linear relationship with the pumping rate for any given spacing between wells in the doublet. With this information in hand, it was therefore possible to predict the critical doublet spacing and therefore the maximum design spacing for a three-doublet system.

Since the physical constraints of aquifer characteristics and site parameters limited the assumed distance between the injection well and the extraction well in a doublet to approximately 1,000 feet (304.8 m) and to pumping rates of approximately 150 gpm (9.5 l/s), a three-doublet system was not sufficiently broad to prevent seawater intrusion through the study area. Therefore, it was necessary to go to a greater number of doublets. The additional I/E doublets raised the question of whether or not the critical doublet spacing will change with the addition of doublets. The computer simulation model was utilized to determine the relationship between critical doublet spacing and number of doublets at any particular pumping rate. The analysis indicated that there is a reduction in critical doublet spacing with the increased number of doublets.

Evaluation of the computer simulation data showed that the critical spacing between doublets is inversely proportional to an exponential of the number of doublets. For the particular case assumed, that of wells separated a distance of 1,000 feet (304.8 m) in each doublet, pumping at 150 gpm (9.5 l/s), it was determined that, for nine doublets, the critical spacing between doublets is approximately 1,800 feet (585 m) and for seven doublets, the critical spacing is approximately 2,000 feet (610 m). Since the actual design spacing between doublets must be less than the critical spacing, and in order that the system be conservatively designed to accommodate any variation which may exist in aquifer characteristics or in the existing hydraulic gradient, a design spacing between doublets was established at 1,000 feet (305 m).

Recommended Location of Injection/Extraction Doublets

Taking into account the physical and cultural characteristics of the study area, together with the design criteria established above for location of the doublet system, the recommended locations of I/E doublets were determined. The system requires a total of nine I/E doublets, the total capacity of which would be approximately 2.0 million gallons per day (mgd) (7.6 × 10^6 l/d). Because of the conservative approach taken in design of the system, a certain amount of latitude was available for use in final location and construction of the doublets.

DESIGN AND CONSTRUCTION

Following the analysis of the computer simulation, the seawater intrusion barrier system was designed. The system includes a total of nine doublets (18 wells), as shown on Figure 12, and 37 monitor wells. Eight existing monitor wells were incorporated into the system together with the initial injection/extraction well (test well).

Monitor Wells

Three types of monitor wells were designed for the injection/extraction well system; shallow wells ("S" series), mid-depth wells ("M" series) and deep wells ("D" series). Monitor wells were designed and located to serve as monitoring points for ultimate operation of the system and as test holes to obtain pertinent information on formation thicknesses and material types for design of the injection/extraction wells.

The "D" series wells were planned to monitor water levels and water quality in the 185-foot aquifer and to provide a supply of fresh water for future possible injection purposes. Each well is designed with 8-inch (20.32 cm) schedule 80 polyvinyl chloride (PVC) casing, type 304 stainless steel wire-wound type well screen, gravel-packed in an 18-inch (45.77 cm) drilled hole. The annulus from ground surface to a point 5 feet (1.5 m) above the top of the screen is sealed with neat cement grout to protect the lower aquifer from possible contamination by water from the shallow zone.

The "S" series wells were designed to penetrate the 20-foot and 45-foot aquifers. The six "M" series wells were designed similarly to the "S" wells with the exception that the initial drilled hole was planned for an approximate depth of 100 feet (30.5 m) for the "M" wells, while the "S" wells were scheduled to be only about 60 feet (18.3 m) total depth.

Each well in the "S" and "M" series is double-cased and double-screened with a neat cement grout seal separating the 20-foot and 45-foot aquifers. Well design includes 3-inch (7.62 cm) PVC casing and PVC wire-wound-type well screen, gravel-packed in a 12-inch (30.5-cm) drilled hole.

Both cuttings samples and split-spoon samples were specified to be obtained during drilling and each hole was to be geophysically logged and calipered to provide additional information for correlation with the geology of the area and for design of the injection/extraction wells.

Injection/Extraction Wells

In order to provide the facility for future use of each of the injection/extraction (I/E) wells for either injection or extraction, each of these wells were designed identically. These wells also are double-cased, double-screened, with neat cement grout separating the 20-foot and 45-foot aquifers. The casing and screen for the 45-foot zone were designed to be 8-inch (20.32 cm) nominal size to allow installation of pumping or injection equipment. Since the 20-foot zone will be pumped only to control waterlogging, the casing and screen for this zone is 6-inch (15.24 cm).

All casing is Schedule 80 PVC and the screens are type 304 stainless steel, wire-wound, keystone-type screens. Each screen is gravel-packed in a 24-inch (61 cm) drilled hole.

Funding

Since the injection/extraction well system is part of the final treatment and disposal process of the regional wastewater treatment plant supplying the reclaimed wastewater for injection, application was made to the Environmental Protection Agency (EPA) and the California State Water Resources Control Board (SWRCB) for grant funding under P.L. 92-500. After review of the predesign report prepared by the Santa Clara Valley Water District and the report of Geohydrologic Investigation, both EPA and SWRCB approved the project for grant funding. Under this program, EPA provides 75% funding of grant eligible costs while SWRCB provides an additional 12½%.

The bid price of the injection/extraction well system was $400,000. Contracts were signed in late 1975 and construction began in January 1976.

Construction

The project was scheduled for completion by the end of August 1976. However, a three-month delay due to right-of-entry restrictions occurred in late spring and early summer. Consequently, at the time of preparation of this paper, only the monitor wells have been constructed.

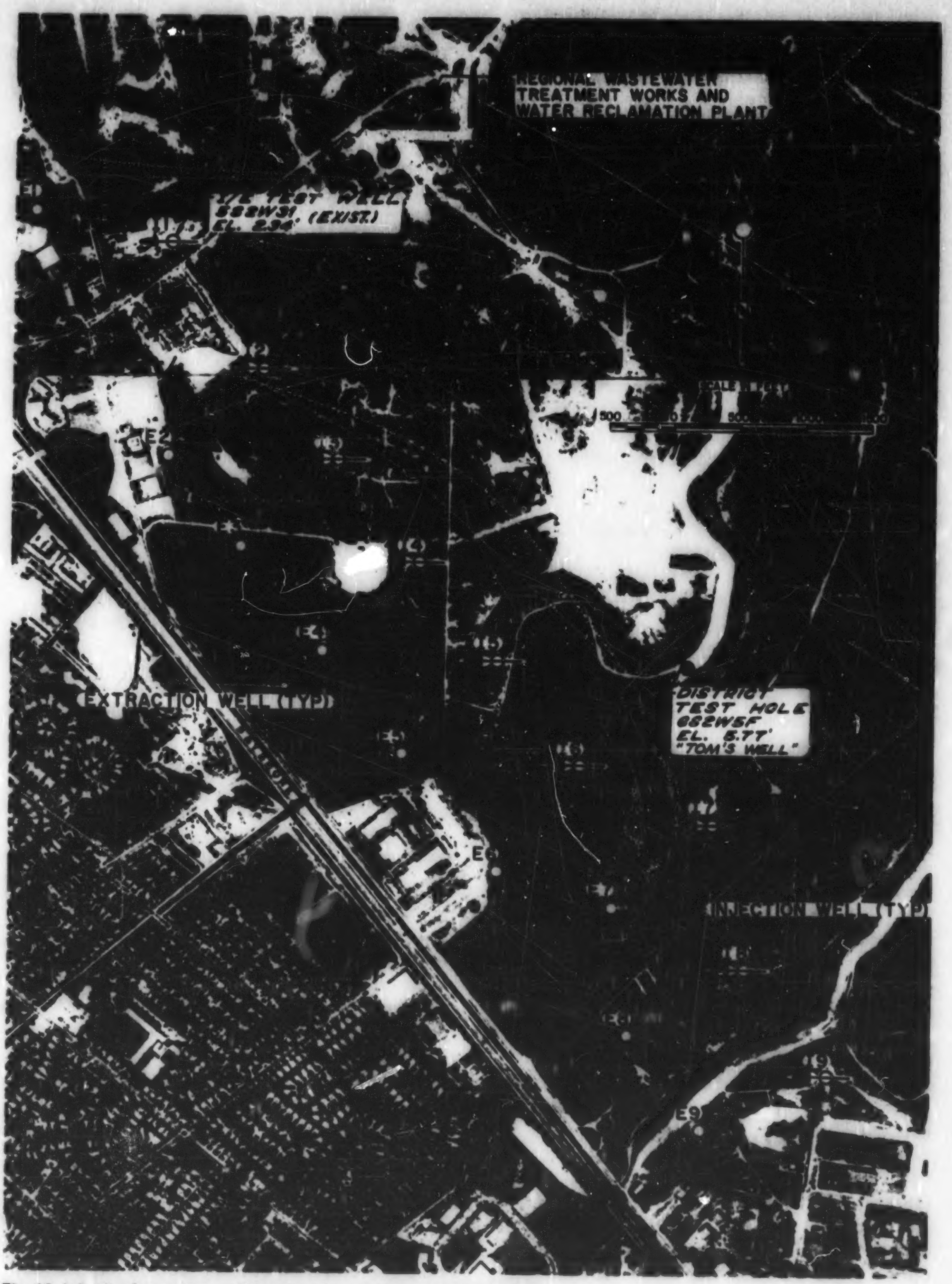

REGIONAL WASTEWATER TREATMENT WORKS AND WATER RECLAMATION PLANT
I/E TEST WELL 6S2W31 EL. 2.34' (EXIST.)
E1
(2)
E2
(3)
(4)
(5)
SCALE IN FEET
500 0 500 1000 1500
EXTRACTION WELL (TYP)
DISTRICT TEST HOLE 6S2W5F EL. 5.77' "TOM'S WELL"
(6)
E6
E7
INJECTION WELL (TYP)
(8)
E8
(9)
E9

Fig. 12. Injection/extraction well locations for seawater intrusion barrier.

It is anticipated that after completion of construction, a supplemental paper will be prepared describing the construction and operation characteristics of the system.

ACKNOWLEDGMENTS

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DISCUSSION

The following questions were answered by N. Thomas Sheahan after delivering his talk entitled "Injection/Extraction Well System—A Unique Seawater Intrusion Barrier."

Q. by Peter Perez. *Would you mention the R & D program to be conducted by Stanford University for transmission of viruses and organics between the injection and extraction wells?*

A. As a long-term goal, the proposed injection/extraction facility will be used for research to determine the feasibility of such a system for reclaiming water for potable uses. A research program has been established by Stanford University, supported by U.S. Environmental Protection Agency research grant EPA-R-804431, in order to answer some of the significant questions as necessary to realize this long-term goal. The major objectives of this research study are as follows:

1. To determine the effects the injected wastewater will have on the chemical, physical and biological quality of the basin and injected waters.

2. To determine the effect injected wastewater will have on the hydrologic characteristics of the aquifer.

3. To seek the optimum quality for injected water which will result in a high-quality basin water and minimum damage to the hydrologic characteristics of the aquifer.

4. To develop generalized mathematical models for describing the movement of water, the changes in hydrologic characteristics, and resulting changes in water quality from wastewater injection in order to make the results of most value for application in other similar projects.

Along these lines, technical report No. 206 dated April 1976 entitled "Preproject Water Quality Evaluation for the Palo Alto Water Reclamation Facility" and the first quarterly progress report dated August 5, 1976 entitled "Groundwater Injection of Reclaimed Water in Palo Alto" have been prepared by the Department of Civil Engineering, Stanford University.

Q. by Jon O. Nowlin. *Was consideration given to the effect on the hydrologic characteristics of the aquitard of using lower-quality effluent as the recharge fluid?*

A. The aquitard allows leakage from the 45-foot zone, into which injection is to take place, to the shallower 20-foot zone. The use of reclaimed wastewater or treatment plant effluent as the recharge fluid may have one of two effects on the aquitard; it may either decrease its permeability or increase its permeability. A third alternative would be no effect at all. Since the project is designed to allow leakage through the aquitard and to control such leakage by pumpage from the 20-foot aquifer, an increase in permeability of the aquitard would have negligible effect on the operation of the project. If, on the other hand, use of reclaimed wastewater caused a decrease in permeability in the aquitard, a reduction in leakage would occur requiring less pumpage from the 20-foot aquifer.

Q. by Stanley N. Davis. *I question the postulated mechanism of seawater intrusion. Are you aware of the large number of abandoned wells including the old Spring Valley Water Company test wells? These are near the salt marsh edge of the bay. They were never adequately filled and subsequently rusted out thus feeding salt water vertically into the underlying aquifers.*
A. Not only the Spring Valley Water Company test wells, but many similar abandoned wells may exist in the study area. All of these wells are potential sources of degradation to the lower aquifers by leakage through the wells from the upper, salinated aquifers. In addition, further westward there is a greater degree of vertical hydraulic communication between the shallow aquifers and the deeper, water-producing zones, which occurs naturally due to the modes of deposition of the sediments. The site for construction of the injection/extraction system was selected so as to preclude any problems such as abandoned wells in its vicinity. There are no abandoned wells located between the injection/extraction barrier and the Bayfront area, the source of saline intrusion into the shallow aquifers in the project area.

Q. by Stanley N. Davis. *Upper zones have a total dissolved solids of about 5,000 ppm. Is this "highly saline"? (See City records of Palo Alto Gold Course test well of about 1964.)*
A. The upper zones, in fact, have total dissolved solids well above 5,000 ppm. For example, the test hole drilled by the Santa Clara Valley Water District in 1972, located immediately east of the injection/extraction system, shows a total dissolved solids of 17,400 mg/l in 1972 and a chloride content of greater than 10,000 mg/l. The Palo Alto Golf Course test well penetrates both the shallow zones and the deeper, fresh-water zones. Perhaps the water-quality analyses from this well reflect a mixture of both salinated ground water and fresh ground water.

Q. by Russel E. Darr. *Do I understand correctly that the injection is in the 40-foot aquifer and the withdrawal is in the 20-foot aquifer?*
A. That is *not* a correct understanding. Both the injection and extraction occur in the same 45-foot aquifer. Injection is planned for the line of wells on the bayward side and extraction is to occur through the line of wells on the landward side. The purpose of the 20-foot aquifer wells is only to allow pumpage to control waterlogging. Injection into the 45-foot aquifer may produce leakage through the aquitard separating the two shallow aquifers and by pumping the 20-foot aquifer, such leakage can be controlled to eliminate any possibilities of waterlogging of the ground surface.

Q. by Dennis Goldman. *Were longer aquifer tests run? Do you feel that a 24-hour test should be extrapolated to imply zero leakage?*
A. This question concerns the zone between the base of the 45-foot aquifer and the lower 185-foot aquifer, herein termed an aquiclude. The total time of pumping for testing of zones on either side of this aquiclude, includin both the pumping tests and the injection tests, amounted to over 96 hours of pumping. Concerning extrapolation of aquifer-test data to imply zero leakage, it is important to consider the scope and purposes of testing as well as the physical geology and other limitations of the test site. Examination of the regional geology of the area shows that the aquiclude changes in character and lithology to the west of the test site. In this area westerly, the aquiclude provides vertical intercommunication between the shallow and deep aquifer due to a pinching out of the clay zones, an increase in permeability due to changes in lithology, or a combination of the two. In addition, the potential for abandoned wells located westerly of the test site which provide vertical intercommunication from the shallow to the deep aquifer through the aquiclude is also present. With continued pumping in a confined aquifer, the area of influence or zone of depression due to such pumping eventually reaches far beyond the area of immediate interest at the test site. Such continued pumpage in the project area would eventually show evidence of leakage from the shallow to the deeper zones. The purpose of the aquifer test, however, was to determine the characteristics of the aquiclude in the vicinity of the test site. Therefore, for purposes of this investigation, the pumping tests performed were adequate to determine the characteristics of the aquiclude in view of both the project requirements and the physical geology of the site.

Q. *You stated that injection of wastewater would increase head and prevent intrusion. What was the head in the upper aquifer before injection? How much can the original head be increased?*
A. The head in the 45-foot aquifer at the injection/extraction well prior to injection was at an elevation of −3.23 mean sea level. It was determined that the original head can be increased by a maximum of 20 psi, or approximately 46 feet, without creating any potential for heaving or leakage around seals in the well.

Q. *Recharge and subsequent extraction has no benefit over salt-water intrusion if the water cannot be used. Do you agree?*
A. I do not agree. Salt-water intrusion has the potential of degradation for the entire ground-water aquifer under heavy pumping conditions in the Palo Alto area, and is therefore detrimental. Recharge and subsequent extraction have the benefit of controlling seawater intrusion as well as providing utilization of the recharged and extracted water for industrial and agricultural purposes downstream of the project area.

Q. *Would you save pumping costs if you stopped seawater from coming into the bay if the Corps of Engineers were to build a locks and dam under Golden Gate Bridge?*
A. This may be a tongue-in-cheek question. Even if the Corps of Engineers controlled the water quality within the bay and maintained fresh water in San Francisco Bay, the existence of salt evaporation ponds in the vicinity of the project site present the potential for further salination of shallow aquifers through leaching from these salt ponds. Therefore, such a project would still be required to prevent salination of the shallow aquifers and to provide protection of the deeper water supplies.

A Nonstructural Approach to Control Salt Accumulation in Ground Water[a]

by Otto J. Helweg[b]

ABSTRACT

One of the more subtle and dangerous areas of pollution is that occurring in ground water from normal irrigation practices. The pollution from salt buildup is presently one of the unsolved problems in managing stream-aquifer systems. This paper presents several strategies that offer a possibility of controlling this salt buildup. For example, instead of applying irrigation ground water near the site of the well, the ASTRAN Method transfers it downstream to be applied on land where the ground water is a lower quality thereby controlling the increase in salt concentration. Instead of preventing seepage loss in delivery canals, the percolating water is used to maintain ground-water quality. Finally, timed releases of return flow remove salts without exceeding surface-water quality constraints.

INTRODUCTION

As is often the case, the most obvious need receives attention first even though it may not be the most important; consequently, the visible problem of surface-water pollution has been emphasized. When the efforts of the NWWA pointed to the equal, if not greater, need to address ground-water pollution, the emphasis has been on dramatic changes such as oil spills, wastes from feedlots, etc., instead of more subtle sources of ground-water degradation. For example, normal irrigation procedures slowly but steadily increase the concentration of total dissolved solids (TDS or salt) in ground water. The process, as shown in Figure 1, works like this. An amount of irrigation ground water (say 1 m^3) with a TDS of 500 mg/l, is applied to the land. One-half of the water (but hardly any of the dissolved solids) is evapotranspired (used by the plants). This means that one-half of a cubic meter of water with a TDS of 1000 mg/l will percolate down to the water table and increase the salinity of the ground water. The natural recharge of the aquifer and lateral ground-water movement is not rapid enough to move the more saline water out so the concentration of salts increases until the ground water and/or land becomes unusable. Before the land was farmed, nature had reached an equilibrium (or nearly so) whereby the discharge of salts via lateral movement

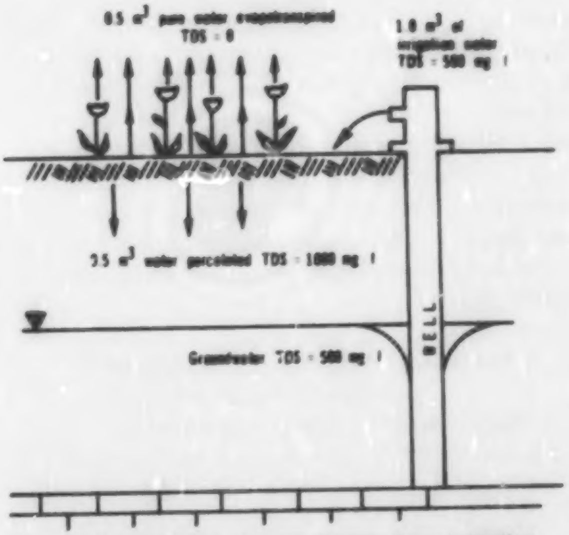
0.5 m^3 pure water evapotranspired
TDS = 0
1.0 m^3 of
irrigation water
TDS = 500 mg/l
0.5 m^3 water percolated TDS = 1000 mg/l
WELL
Groundwater TDS = 500 mg/l

Fig. 1. The degradation of ground water from irrigation.

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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equaled the increase in concentration from natural vegetation; moreover, the applied water was rain which had a very low salt content. Though measurements have confirmed this process, as yet there is no economic procedure to control it, notwithstanding, many proposals which may retard degradation or delay it for a period of years.

STANDARD METHODS OF SALT CONTROL

A few forward-looking organizations are addressing the problem of ground-water degradation. Among these is the Colorado River Water Quality Improvement Program. The Program Officer has listed the various possibilities in water-quality planning and management (Maletic, 1974). These are given in Table 1.

The management techniques presented herein would appear to fall under Category III 3, Ground-Water Management, since they involve selective pumping, application, and release of ground water. They might also be classified under Category IV, River System Management. These techniques are not presented as the ultimate solution to ground-water salinity problems, but a potentially viable alternative for consideration among the many other alternatives. The challenge to water planners and managers is to find the optimal mix of alternatives that will meet water-quality goals in the most beneficial way.

ACCELERATED SALT TRANSPORT (ASTRAN) METHOD

Many stream-aquifer systems may be conducive to application of a potentially cost-effective salinity management technique referred to as the Accelerated Salt Transport (ASTRAN) Method (Helweg and Labadie, 1976). The usual irrigation practice is to apply ground water on fields near the well supplying the water. The ASTRAN Method, however, encourages the application of ground water on downstream fields (via canals or pipes) instead of on nearby fields. That is, fields proximate to a pumping well should be irrigated, to at least some extent, by water from upstream wells. Salts in the pumped water can therefore be transported downstream at a faster rate than would occur naturally through flow in the saturated zone; hence, the name Accelerated Salt Transport (ASTRAN) Method. The slow movement of ground water tends to cause an accumulation of salts from normal irrigation practice, since drainage water adds salt to the aquifer at a faster rate than it can be naturally transported downstream. The idea behind this technique is to simply augment the natural process by transporting pumped water downstream via a surface distribution system of some kind.

The management technique is illustrated in Figure 2. This diagram depicts an increasing salt concentration downstream, since many stream-aquifer systems have this characteristic. Such a condition would not seem to be necessary for successful application of the ASTRAN Method. It is, however, required for application of the management algorithm presented in this report. Appropriate modification of the algorithm could be carried out, in case this condition does not exist.

Table 1. Major Categories of Salinity Control

I. POINT SOURCES
1. Desalt
2. Divert/Evaporate
3. Divert/Special Use
4. Plug Wells
5. Deep Injection

II. NATURAL DIFFUSE SOURCES
1. Collect/Desalt
2. Collect/Evaporate
3. Collect/Special Use
4. Watershed Management
a. Vegetative conversions
b. Forest management
c. Structural measures
d. Water harvesting
e. Reduced sediment production
5. Phreatophyte Control
a. Control of spread
b. Replacement vegetation
c. Antitranspirants

III. IRRIGATION SOURCES
1. Improved On-Farm Irrigation Use
a. Irrigation scheduling
b. Improved farm irrigation systems
1) Pipes and lining
2) Automation
3) Advanced systems
2. Improved Water Conveyance Systems
a. Piles, lining
3. Ground-Water Management
a. Water-table control (drainage)
b. Selective pumping
c. Ground-water recharge
4. Return Flow Management
a. Collect/desalt
b. Collect/special use
5. Evaporation Suppression

IV. RIVER SYSTEM MANAGEMENT
1. Alteration of Time Pattern of Streamflow
2. Alteration of Time Pattern of Saline Discharges

V. DILUTION
1. Augmentation
a. Weather modification
b. Geothermal resources
c. Desalting
d. Wastewater reclamation
e. Conservation practices
2. Importation

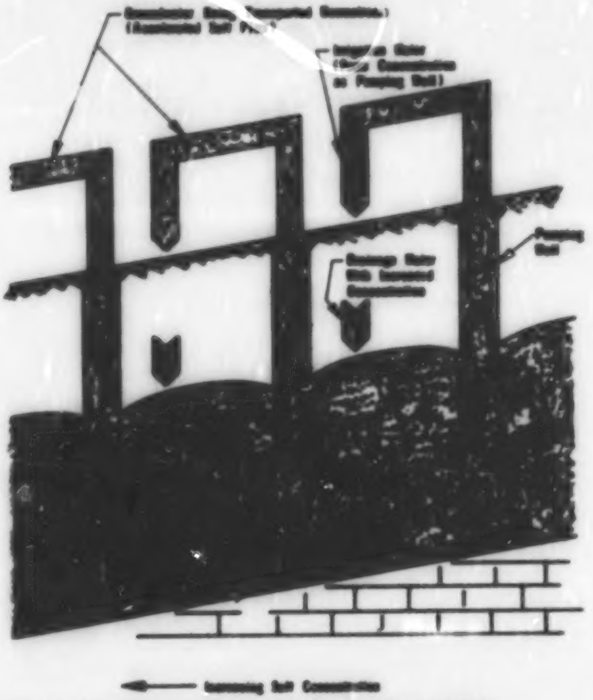

Fig. 2. A schematic diagram of the Accelerated Salt Transport (ASTRAN) Method.

Quantifying the ASTRAN Method entails finding out how much water should be taken from each source and where it should be applied. It is certainly possible to pump more than needed and maintain a hydrologic balance by artificial recharge if the constraints and economics so demand. If water quality were not a consideration, one could just apply a subset of Linear Programming (LP) called the Transportation Problem (see Wagner, 1975).

The Transportation Problem is designed for a situation where there are supply points (sources), demand points, and costs to transport the items between each source and demand. Figure 3 illustrates this situation. There is a least cost solution to this problem which can lead to a solution to the ASTRAN Method. Figure 4 shows the Bonsall subbasin in California, where the ASTRAN Method was tested by simulation models. The subbasin was divided into four sections, each one being both a source and demand. Additionally, two other sources were available; ground water from upstream basins (USGW), and imported State Project water (SPW). There was also an additional demand and that was for water needed downstream or exported out of the subbasin (EXP).

The amount of water demanded is set by the crops and irrigated land. The amount of water available is determined by the aquifer size and

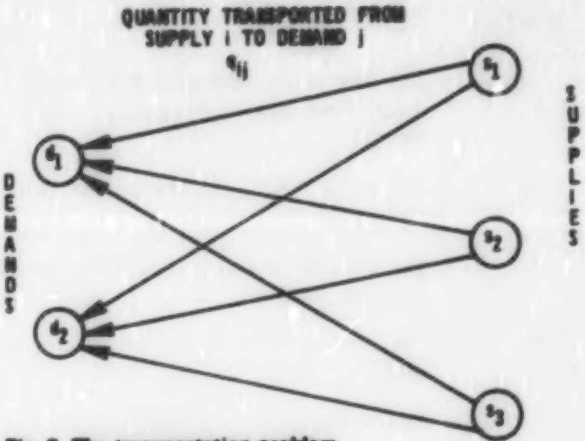

Fig. 3. The transportation problem.

characteristics. The costs for transporting water between sources and demands is calculated by water costs (as for imported water), pumping costs (for ground water), and the distance it needs to be transported (costs of canals, etc.). Mathematically, the problem is formulated as follows:

$$\min_{q_{ij}} \sum_{i=1}^{n} \sum_{j=1}^{m} c_{ij} q_{ij}$$

$$i=1, \ldots, n$$

$$j=1, \ldots, m$$

subject to:

$$\sum_{j=1}^{m} q_{ij} < s_i, \; i=1, \ldots, n$$

$$\sum_{i=1}^{n} q_{ij} > d_j, \; j=1, \ldots, m \qquad (1)$$

$$q_{ij} > 0, \; i=1, \ldots, n; \; j=1, \ldots, m$$

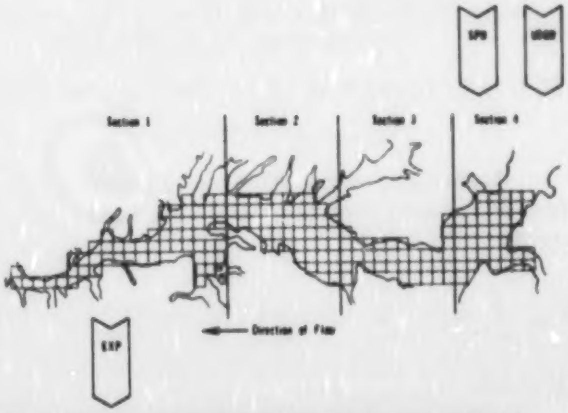

Fig. 4. Supply and demand points for Bonsall subbasin.

in which:

c_{ij} = the cost of transporting water from source section i to demand section j ($/AF);

q_{ij} = the amount of water transferred from i to j (AF/yr);

s_i = the amount of water available at source section i (AF/yr);

d_j = the amount of water needed at demand section j (AF/yr).

As mentioned above, the Transportation Algorithm is only the basis of a solution strategy because, as yet, water quality has not been considered. The TDS of each supply is different and in order to control ground-water degradation the applied irrigation water must be of sufficiently higher quality to maintain mass balance of salts. Consequently, the algorithm needs an additional water-quality constraint. The problem is to find the quality of the applied irrigation water needed in order to maintain ground-water quality. An initial approximation is to assume a static situation and use the leaching formula:

$$D_w = \frac{EC_{dw}}{EC_{dw} - EC_w} \times ET \qquad (2)$$

in which:

D_w = the depth of the supplied irrigation water for leaching (cm);

ET = the evapotranspiration or consumptive use (cm);

EC_{dw} = the electroconductivity of the drainage water percolating past the root zone (micromhos/cm²);

EC_w = the electroconductivity of the applied irrigation water (micromhos/cm²).

EC is easily found from TDS by the relation,

$$TDS = -2 + 0.683\ EC.$$

Rearranging terms, assuming six sources and four demands of irrigation water as in Figure 4 yields:

$$C_{mj} \sum_{i=1}^{6} q_{ij} - \sum_{i=1}^{6} q_{ij} C_i \geq C_{mj} ET_j, j=1, \ldots, 4 \qquad (3)$$

in which:

C_{mj} = upper bound on average TDS concentrations of the drainage water at section j (mg/l);

C_i = average TDS concentration of irrigation water from source section i (mg/l);

q_{ij} = amount of water transferred from source section i to demand j (AF/yr);

ET_j = evapotranspiration at section j (AF/yr).

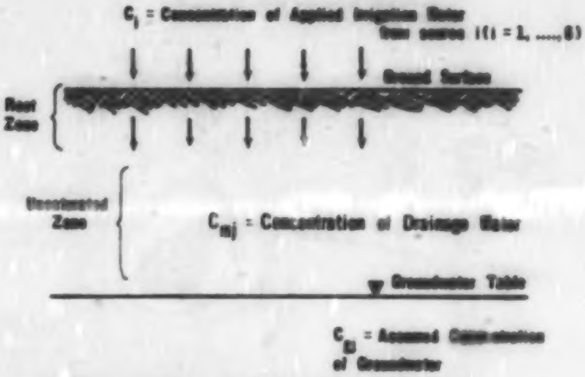
C_i = Concentration of Applied Irrigation Water from source i (i = 1, ..., 6)
Ground Surface
Root Zone
Unsaturated Zone
C_{mj} = Concentration of Drainage Water
Groundwater Table
C_{Gj} = Assumed Concentration of Groundwater

Fig. 5. Water-quality schematic of demand node j.

Figure 5 shows the locations of the various waters for formula (3).

The situation is a little more complex than Figure 5 because the aquifer is not static as assumed by the leaching formula. Ground water is flowing in and out of the aquifer from the boundaries plus water is entering or leaving the aquifer from rain or via the river. For the Bonsall basin, it was found that the water-quality constraint could be up to 300 mg/l greater than the actual ground water and maintain a near stable salt balance. Figure 6 shows the long-term solution for Bonsall basin and Figure 7 shows the improvement of the downstream portion of the basin the ASTRAN Method would have had if it had been applied back in 1966.

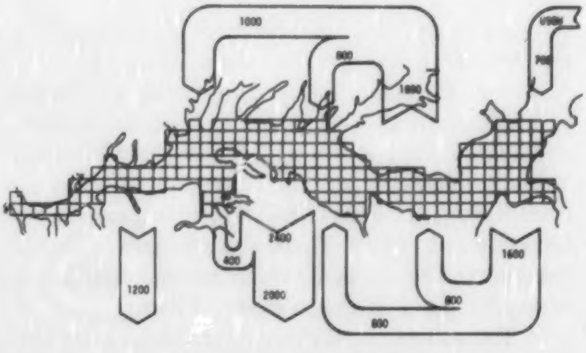

Fig. 6. Long-term solution for Bonsall subbasin (numbers are water transfers in acre-feet/year).

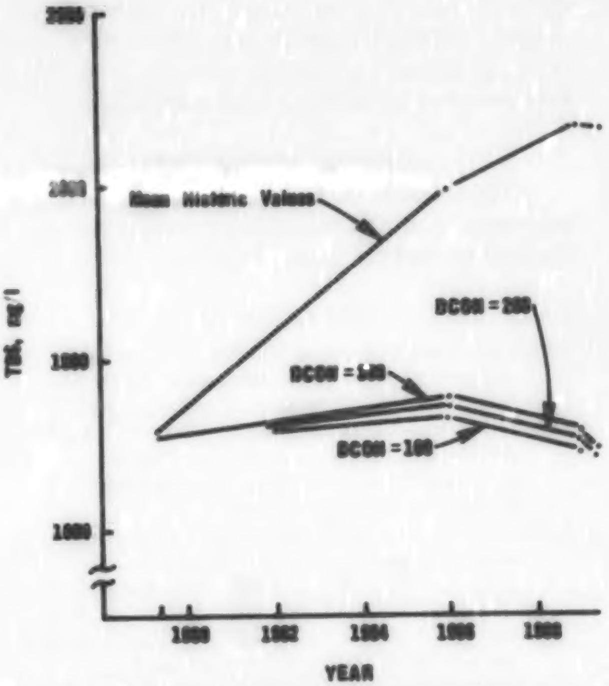

Fig. 7. Ground-water quality, comparing the historic values with the ASTRAN Method (DCON is the difference between ground-water quality and drainage-water quality).

CONTROLLED SEEPAGE RECHARGE

Depending on the situation, the ASTRAN Method may call for artificial recharge to meet the water-quality and quantity constraints. Often this can be accomplished by Controlled Seepage Recharge (CSR) that is allowing seepage in delivery canals purposely by so designing the water delivery system that the seeping water not only helps to maintain a hydrologic balance, but a water-quality balance. Figure 8 shows how this might be done.

Most irrigation systems' designers consider seepage a loss, and line delivery canals to prevent this. Though this may often be desirable, lining canals is not always called for. For example, Konikow and Bredehoeft (1974) showed that unlined canals were, in fact, helping to maintain the ground-water quality of the Arkansas River basin in Colorado. Basically, the canals can be designed for two purposes, water delivery and artificial recharge. It may be that such a multipurpose use of the canal would make the construction of artificial recharge facilities unnecessary.

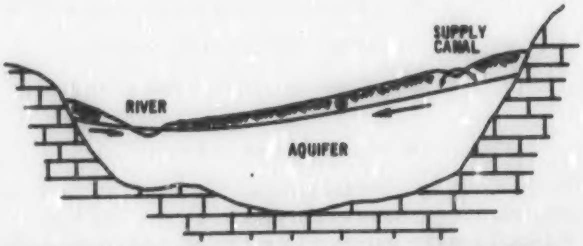

Fig. 8. Controlled seepage recharge from an unlined ditch.

TIMED GROUND-WATER RELEASE

It is not yet clear what the impact of the Federal Water Pollution Control Act (PL 92-500) will have on irrigation return flow; but if a river is to maintain a certain water quality such as a controlled TDS, the irrigation return flow from the drains (which cause the river to exceed this during part of the year) may be prohibited. Normally, irrigated areas are in climates where the Summer river flows are low; consequently, the relatively saline water returning to the river from irrigated lands have a much higher effect on the surface-water quality than they would during the Winter or Spring months. An example of Timed Ground-water Release (TGR) is operating tiled places so that the outfalls into the rivers can be shut during the Summer months and then opened during months of high flow. Moreover, if pumping is called for to export saline water, this can be accomplished during these same periods of high flows.

For example, if the TDS of a river were constant, and the maximum allowable TDS were 800 mg/l, a timed ground-water release policy as shown in Figure 9 during the months of January through March would not exceed the surface-water-quality standards. Often, however, the TDS during

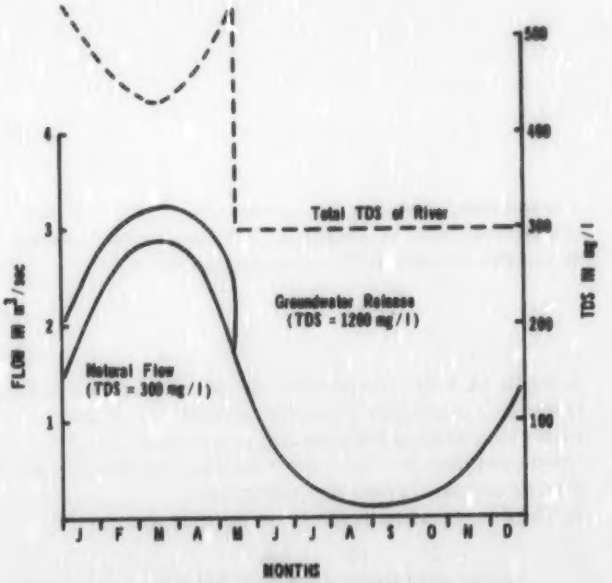

Fig. 9. The change in river water quality from timed ground-water releases.

the periods of high flow is lower than the low flows, so this would show an even more favorable picture than the one illustrated. Of course the rate of ground-water release need not be constant but could be matched to the quantity of the natural flow. Some work along these lines has been done in Australia in the Murray River basin (Williams, 1972).

CONCLUSIONS

The main point of this paper is that often water-quality objectives can be approached by including nonstructural measures in the over-all management program. These measures have the advantage of requiring low initial investment costs. For example, one estimate for decreasing the rate of ground-water degradation, but not halting it, in the Bonsall area was $3,900,000 (Santa Margarita-San Luis Rey Watershed Planning Agency, 1973) while the ASTRAN Method could completely control degradation at roughly 10% of that figure (Helweg and Labadie, 1976). Nonstructural measures provide greater future flexibility in that future options have not been decreased by constructing fixed facilities.

However, several words of caution are needed. As with many water problems, the technical solution is much easier than the political. For the CSR or the TGR methods to work effectively, the whole river basin should be coordinated and this is difficult when there are many ditch companies or irrigation districts. Nevertheless, the potential gains of nonstructural measures to control salt accumulation seem to warrant their implementations.

ACKNOWLEDGEMENTS

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DISCUSSION

The following questions were answered by Otto J. Helweg after delivering his talk entitled "A Nonstructural Approach to Control Salt Accumulation in Ground Water."

Q. by Don Lundy. *The ground-water flow equation presented in one slide is written for one-dimensional flow in the x-direction. Subsequent maps and sections indicate component flow in at least one more direction (horizontal). How do you justify flow in a y-direction?*

A. The flow equation as shown on the slide was:

$$\frac{\partial}{\partial x_i}(T_{ij}\frac{\partial h}{\partial x_i}) = S\frac{\partial h}{\partial t} + W(x,y,t)$$

It should have been:

$$\frac{\partial}{\partial x_i}(T_{ij}\frac{\partial h}{\partial x_j}) = S\frac{\partial h}{\partial t} + W(x_i,t)$$

in which:

x_i = the spatial cartesian coordinates (i=1,2)(L);
T_{ij} = the second order transmissivity tensor (L^2/T);
S = the storage coefficient (dimensionless);
h = the piezometric surface above some datum (L);
W = the volume flux per unit area (L/t).

So the model was two-dimensional; however, because of the long, narrow geometry of the river basin, we could have modeled it with a one-dimensional model. The reason we didn't was because the USGS had already calibrated a two-dimensional quantity model and it seemed reasonable to utilize the work they had already completed.

Q. by Leonard Konikow. *What value of dispersivity did you use and how sensitive was the model to this parameter?*
A. Our final values for BETA (the characteristic length) was 100 and the value we used for DLTRAT (the ratio of transverse to longitudinal dispersivity) was 0.3. We did an informal sensitivity analysis with these two numbers and the model was not sensitive to them. In fact, with BETA equal to zero (i.e., dispersivity equal to zero), we still got good results.

Q. by Robert O. Thomas. *Is there any work being accomplished on the accelerated transmittal of a contaminant—at least transmittal evidenced by odor, etc.? I refer specifically to phenol movement of over 10 miles in a week in a ground-water body moving in the low hundreds of feet per year. This occurred in the lower San Gedowel Valley, California in the early thirties.*
A. Not to my knowledge.

Q. by Robert O. Thomas. *Does not properly managed conjunctive use, using surface water for both recharge and supplemental applied water, have a large positive effect on maintenance of ground-water quality?*
A. How do you "properly manage conjunctive use"? We are proposing the ASTRAN Method as a "properly managed conjunctive use" program. However, the ASTRAN Method can be applied where there is no surface water available to be imported into the river basin. The problem is that there is often not enough good quality surface water to go around.

Q. by G. H. Hendricks. *Would you briefly discuss rainfall amounts relative to leaching? What are the limitations?*
A. The precipitation average at the upper end of the basin is about 13 inches. We did not consider the leaching effect of this rain in relation to the root zone because we were investigating the water quality on a larger scale, i.e. the ground-water table. The precipitation was included in the simulation model, however. There have been studies on leaching from precipitation, but the information was not pertinent to our study.

Q. *Who in USGS developed the model that incorporates water quality? What division of USGS?*
A. The simulation model was developed by Bredehoeft and Konikow of the Water Resources Research section of the USGS at Denver. Their work was based on research done at Colorado State University by Redell and Sunada. This model solves both the flow equation and the transport equation and is one of the few working models that does so.

Q. by John E. Mann, Jr. *Do you know the magnitude of TDS pick-up shown by the Imperial Valley drains?*
A. The average electroconductivity (EC) of the irrigation water from the Colorado River is 1300 micromhos/cm and the drainage water from the tiles ranges from 2300 to 3900 micromhos/cm. If you are talking just about surface runoff, the EC increases from 5 to 10%.

Q. by W. B. Wilkinson. *Were sensitivity tests applied to the simulation model and if so, which of the aquifer's properties proved to be the most critical with respect to decision making? How were the dispersivity values used in the model determined?*
A. Concerning the flow portion of the simulation model, initial water levels were the most sensitive factor, followed by transmissivity and storage coefficient. Informal sensitivity tests were used, but the parameters, such as transmissivity, were not formally optimized.

The dispersivity values were taken from USGS values measured in similar systems. Since they proved to have very little effect on the results, there was no need to spend money testing the aquifers in the basin.

Q. by L. A. Swain. *You assume (in model) that recharged water mixes uniformly throughout entire vertical thickness of aquifer. Do you have any monitoring data to support this idea rather than the recharge water being stratified atop the aquifer water?*
A. We do not have any monitoring data because there is no artificial recharge presently being done. If artificial recharge would be used, the design of the pits (or wells) would have a great effect on the mixing. Because of the geometry of the aquifer though (i.e., shallow with not much layering), we would not expect stratification to be a major problem. Remember, though, the ASTRAN Method does not necessarily need artificial recharge; artificial recharge would only be used if the decision maker wanted to so improve the ground-water quality that the normal transfer of irrigation water was insufficient.

Q. by Donald Runnells. *Does the ASTRAN model consider any chemical reactions such as precipitation of salts, ion exchange, etc.?*
A. No. Investigators at the University of Arizona and others have modeled the chemical reactions in the root zone and vadose zone, but doing this would have greatly complicated the simulation model and yielded little if any benefit. Remember that we are managing water quality on a large scale so the models are appropriately (we trust) simplified.

Q. by K. Childs. *Does the import of water address the problem or simply the symptom? Is this not using dilution as a solution to pollution?*

How does your suggested methodology address the total quantity of salts in the hydrologic regime?
A. To answer the first part, we feel that the importation of water can be merely diluting saline water, but using the ASTRAN Method would prevent this. What the ASTRAN Method does is to apply the imported water at the best place in order to maintain the appropriate water-quality gradient between drainage water and applied water. The point of application would also assist in moving the salts downstream. Moreover, as stated above, imported water is not necessary to implement the ASTRAN Method.

The simulation model calculated the mass balance of salts, though at the time we used it there were some inaccuracies that have subsequently been corrected by Knokow at the USGS. Consequently, the ASTRAN Method not only controlled water quality (as measured by total dissolved solids), but caused a salt balance in the system also. Obviously water quality and mass balance of salts are connected, but the decision maker can look at both in deciding to what degree he wants to control the ground-water quality.

Quantifying the Natural Flushout of Alluvial Aquifers[a]

by John S. Fryberger and William H. Bellis[b]

ABSTRACT

A plume of contaminated water may extend downstream for several miles in an alluvial aquifer. Hydrologic forces acting on the aquifer generally tend to dilute the contaminant downstream from the source. After the pollution has been stopped, how rapidly will these hydrologic forces flush out the contaminant? The answer to this question may be calculated by using the mass-balance equation herein developed.

This mass-balance mathematical model incorporates all the hydrologic forces acting on the alluvial aquifer that affect the concentration of the contaminant. Methods are presented to define and quantify each of the hydrologic forces. These forces include (1) present quantity and quality (with respect to contaminant) of water in the alluvium and the change in the quality downstream, (2) quantity and quality of ground-water inflow, (3) quantity and quality of flood inflow, (4) quantity and quality of base flow, (5) quantity of recharge from precipitation, and (6) quantity of loss from evapotranspiration.

The equation is first balanced to agree with past conditions and observed field data and is then used to predict future quality changes after the pollution is stopped.

The method is applicable primarily to situations in which the pollution has been taking place over a long period of time and the water quality in the alluvium has reached equilibrium at any given point. Modifications may be possible to permit use of the general approach to short-term or slug-type pollution events.

INTRODUCTION

Along the drainage systems of the Red River of Oklahoma and northern Texas and the Arkansas River of Oklahoma and Kansas are significant natural salt-water springs and seeps which discharge approximately 20,000 tons of salt (NaCl) per day into these river systems.

This salt-water inflow has a significant adverse effect on the quality of the surface water as well as contaminating the river alluvium for hundreds of miles downstream from these natural springs. The Corps of Engineers has the responsibility for developing methods of control, which in most cases will involve the collection and disposal of brine in off-stream evaporation reservoirs, thus essentially stopping the salt from entering the river systems. However, the flushout of the salt from the alluvium downstream from the source areas and the effect of such flushout on surface-water quality are of major concern.

As a result of this interest a mathematical model was developed for the Corps of Engineers which describes the hydrologic forces that control the flushout process. This model was applied to the Red River and several tributaries in Oklahoma and Texas with apparent realistic success. The concepts and procedures developed during the course of this study work may have general application for the prediction of the long-term impact and fate of pollution plumes in alluvium.

The application of the mass-balance model is presently limited to those areas where the source of pollution has been active for sufficient time to allow a state of equilibrium to be established for the decay of pollutant concentration downstream from the source. Modification of the model will be required to define the future flushout of plumes not yet at equilibrium with the flushout forces.

BASIC CONCEPT

The fundamental principles of a hydrologic mass-balance equation can be expressed by defining the volume of water and the chemical concentration for each parameter as follows:

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

[b] Vice President and Hydrogeologist; and Hydrogeologist, respectively, Engineering Enterprises, Inc., 1225 W. Main, Norman, OK 73069.

$$VC_o + IC_i - LC_r = VC_r\,, \qquad (1)$$

where:

V = original and ending volume of water,

C_o = original concentration of the contamination in V,

I = volume of water being added (inflow) to the original volume,

C_i = concentration of the water added,

L = volume of water removed after mixing (equal to volume added),

C_r = concentration of water after mixing (applied both to the water removed and water remaining), and

VC_r = result in which the volume is the same as the original volume but the concentration has changed.

The inflow water, I, with concentration, C_i, is mixed with the initial volume of water with a concentration, C_o, to form a new mixture with a concentration of C_r. Then the quantity, L, which is equal to the quantity, I, but having a concentration of C_r, is removed, leaving the original volume, V, with a concentration of C_r.

Because there are two components to each factor, it is possible to write two equations and, therefore, solve for two unknowns.

$$V + I - L = V, \qquad (2)$$

$$VC_o + IC_i - LC_r = VC_r. \qquad (3)$$

This mass-balance principle is herein expanded to define the hydrologic forces acting on an alluvial aquifer. These forces are depicted in Figure 1.

THE MODEL

General

Expansion of the mass-balance concept to describe the hydrologic forces acting on an alluvial aquifer results in the following equation:

$$QC_q + GC_g + FC_f - BC_r + P - E + RC_r \qquad (4)$$

where:

Q = original quantity of water stored in the alluvium per unit distance,

G = quantity of ground-water inflow from bedrock into the alluvium,

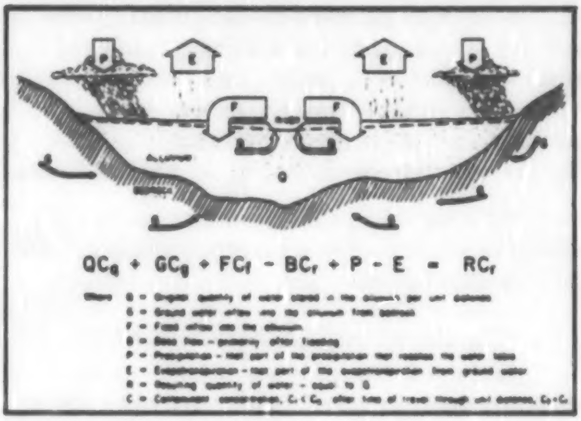

Fig. 1. Schematic of alluvial flushout equation.

F = quantity of flood inflow that mixes with the alluvial water,

B = quantity of base flow from the alluvium into the surface stream,

P = quantity of precipitation that mixes with the alluvial water,

E = quantity of water lost from the alluvium by evapotranspiration,

R = quantity of water remaining in the alluvium, which over a long term is equal to Q, and

C = contaminant concentration of each respective volume of water in milligrams per liter (mg/l).

Concentrations of P and E are assumed to equal 1 and are not shown in the equation.

In general, the model is applied to field situations by calculating and adjusting the values of each component to match observed field conditions. Future predictions may then be made by changing the value of the key input factor to its future value.

Detailed discussions of the mechanics of applying the model follow later. The basic steps are:

a. Determine from field observations the present decay rate of the contaminant downstream from the source, $C_q - C_r$, per unit distance. This change in the alluvial water concentration, $C_q - C_r$, takes place over a calculable period of time, t.

b. Over the time period, t, calculate the values of GC_g, P and E.

c. The values for F and B may be calculated using simultaneous equations.

d. After all values of the factors have been defined so that the equation balances using the observed values of C_q and C_r, adjustments can be made if the values of any parameters appear unreasonable. An independent check is available for the value for F to determine its reasonableness.

e. Future values for C_r can then be made by changing one of the input parameters such as C_f to an estimated future value.

CALCULATING EACH COMPONENT

As a given volume of water moves slowly downstream in the alluvium, the concentration of the contaminant decreases as a result of the hydrologic forces acting on that volume. It is assumed that although water is added and removed, the volume of water remains constant over the long term. However, the quality changes with distance downstream from the source and can be determined by field testing. By calculating the time for water in the alluvium to move a unit distance, and because the change in the quality over that distance is known, $C_q - C_{r1}$, the change in quality can be related to time. The factors that cause the change in quality over that same time period can then be calculated and the equation established.

QC_q and RC_r

The original quantity, Q, and resultant quantity, R, are equal and represent the volume of water in an average one-mile stretch of the river along the study reach. As this volume of water moves downstream the concentration of the pollutant decreases. The average decrease, $C_q - C_r$, per mile, or some other unit distance, is established by the existing decay gradient determined from test drilling, as illustrated in Figure 2. Note that, in the example shown in Figure 2, the river has been divided into three flushout reaches on the basis of changing hydrogeologic conditions along the river.

The time required for the alluvial water to travel one mile is then calculated. The velocity of the ground water is calculated using the following equation:

$$v = \frac{KI}{7.48\ Pe} \qquad (5)$$

where:

v = velocity in feet per day,

K = permeability in gpd/ft²,

I = hydraulic gradient,

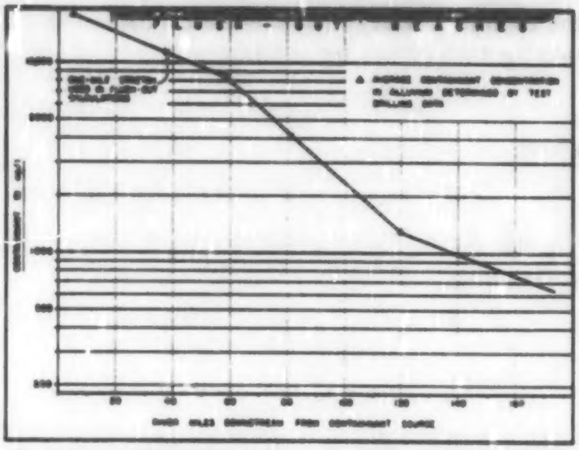

Fig. 2. An example of natural present decay of contaminant downstream from source.

Pe = effective porosity, a value less than total porosity, P_t, due to friction and viscosity—$Pe = P_t/2$ is suggested,

7.48 = constant.

The time of travel, t, for one mile is then determined by:

$$t \text{ in years} = \frac{5280 \text{ ft/mi}}{(365 \text{ d/yr})(v \text{ ft/d})}. \qquad (6)$$

GC_g

The quantity of ground-water inflow, G, into the alluvium from the adjacent bedrock can be determined by standard methods. The average ground-water gradient toward the river along the reach being studied is obtained from water-level-contour maps. The permeability is averaged from pumping tests or estimated based on a knowledge of the formation types. The quantity, G, is estimated using the permeability, gradient, area of inflow into the alluvium and the time of travel determined from equation (6). This represents the quantity of ground water that mixes with the original quantity, Q, in the alluvium during the time that Q water travels one mile (the unit distance chosen for this example).

$$G = \frac{365\ KIA\ (t)}{325{,}829} \qquad (7)$$

where:

G = quantity of ground-water inflow in acre-feet in t, time,

K = permeability in gpd,

I = ground-water gradient in the bedrock,

A = area of inflow = (5280 ft) (depth of alluvium) (2),

t = time in years from equation (6),

365 = days per year,

325,829 = gallons per acre-foot.

The concentration of the pollutant in the ground water in the bedrock, C_g, is determined from chemical analyses of bedrock-water samples. If no pollutant is present, C_g is assumed to equal 1.

P

The quantity of precipitation, P, that infiltrates into the alluvium along a one-mile stretch of river is a fraction of the total precipitation, P_t, and is dependent on several factors such as soil type, vegetation, slope, rainfall intensity and duration, and temperature. It is generally not practical to define this quantity precisely. An estimate ranging from 5 to 30 percent of the total precipitation can be made by judging the effects of the controlling factors. P is calculated as follows:

$$P = \frac{J\, P_t\, A\, t}{12} \qquad (8)$$

where:

P = quantity of precipitation mixing with alluvial water in acre-feet,

J = judgment factor, ranging from .05 to .3,

P_t = total average annual precipitation in inches,

A = area of the alluvium in one mile in acres,

t = time in years from equation (6),

12 = correction factor to obtain acre-feet.

E

The quantity of water lost from the alluvium by evapotranspiration, E, is an important factor in the mass-balance equation but is not easily determined. Methods for calculating evaporation are found in Sleight (1917), Hughes and McDonald (1966), Sorey and Matlock (1969), Hellwig (1973), and Gaetwood and others (1950).

Cruff and Thompson (1967) concluded, in a comparative study of different methods for calculating transpiration, that the Blaney and Criddle (1950) method is generally the most practical. The reader is referred to the above references for detailed instructions on the calculation of evapotranspiration losses.

Factors that influence transpiration losses include:

Depth of water table.
Type of vegetation.
Density of vegetation.
Percentage of area covered by vegetation.
Area per mile length of river underlain by alluvium.
Climatological factors.

Factors that influence evaporation losses include:

Temperature.
Wind.
Depth of water table.
Soil types and grain size.

Potential evaporation and transpiration losses can be calculated for each subarea in the one-mile stretch as follows:

Percentage of total area for open running water (river surface).
Percentage of total area where the water table is less than two feet deep.
Percentage of total area where the water table is more than two feet deep.
Percentage of total area vegetated, and corrected for density of vegetation.

The sum of the preceding evaporation and transpiration losses constitutes the total potential evapotranspiration, E_t. The amount of evapotranspiration that affects the quantity of ground water in the alluvium, E, is the total potential evapotranspiration, E_t, less that part satisfied by soil water above the water table. The amount of soil water may be estimated as that precipitation that does not run off or does not percolate to the water table. The percent of total precipitation that runs off is governed by the amount of precipitation, time distribution and intensity of storms, soil type, surface slope, and density of vegetation. Judgement also must be used in estimating the amount of slope wash or other input to the alluvium from runoff from the bedrock-drainage area.

Although the subarea fractions of E_t are calculated in terms of acre-feet, the value of E_t should be expressed in terms of inches over the entire area of the alluvium in the one-mile stretch.

E is then calculated as follows:

$$E = \frac{[E_t - (P_t - P - P_r)]\ At}{12} \qquad (9)$$

where:

E = quantity of evapotranspiration affecting the alluvial water in acre-feet over time, t,

E_t = total potential annual evapotranspiration in inches,

P_t = total annual precipitation in inches,

P_r = estimated amount of total precipitation that leaves the system as direct runoff, in inches,

P = quantity of precipitation that percolates to the water table from equation (8) but expressed in inches,

A = area underlain by alluvium in acres in one-mile stretch,

t = time in years from equation (6).

FC_f

During periods of flooding, considerable amounts of flood water, F, enter the alluvium and mix with the alluvial water. The quality of this water, C_f, is determined from flow-concentration-duration curves or other direct measurements of the concentration of the flood waters.

The quantity of flood inflow, F, over time, t, can be directly estimated from gage height, flood frequency curves, unsaturated thickness, surface profiles across the river, and porosity data. However, this calculation is even more difficult than calculating the other factors and is done primarily as an order-of-magnitude check on the value of F determined using the equation. The quantity of flood inflow can be more easily determined using the mass-balance equation. At this point in the discussion all the factors have been calculated except F, flood inflow, and B, base outflow. Using the two basic equations, these unknowns may be determined as follows:

The quantity-only equation is:

$$Q + G + F - B + P - E = R, \qquad (10)$$

and because Q = R, then

$$G + F + P = B + E \text{ or inflow} = \text{outflow}, \qquad (11)$$

and B = F + G + P − E. Then by substitution for B in the full equation,

$$QC_q + GC_g + FC_f - (F + G + P - E)\,C_r + P - E = RC_r\,, \qquad (12)$$

and solving for F

$$F = \frac{QC_q - RC_r + GC_g - (G + P - E)\,C_r + P - E}{C_r - C_f}. \qquad (13)$$

$B\ C_r$

The quantity of base outflow, B, from the alluvium into the surface flow is extremely difficult to measure. Therefore, this value is determined from the quantity-only part of the equation after other values have been calculated as follows:

$$B = F + G + P - E. \qquad (14)$$

The concentration of the base outflow, C_r, is assumed to average the same as the resulting water concentration, C_r, as developed in the basic concept under equation (1). It should be noted that the actual concentration of the base outflow varies with time and that a more accurate representation would be to express the pollutant removal in terms of tons per year rather than as a concentration. This value represents the quantity of pollutant or base load, BL, that moves from the alluvium into the surface flow as the alluvial water flows downstream a unit distance in time, t.

After all the factors in the mass-balance equation have been initially estimated, it may be necessary to adjust some of the parameters to get the equation to balance and to obtain reasonable values for each factor. Once this is done, then future predictions can be made based on an assumed or known change of one of the factors.

FUTURE PREDICTIONS

The objective of the mass-balance equation is to predict the rate of future flushout after the contamination source has been stopped. The present application of the equation is predicated on the assumption that the hydrologic forces and the contaminant concentration are in equilibrium, and that a future change in a major factor is required to bring about a reduction in alluvial contaminant concentration leading to a new equilibrium.

On the project for which the equation was developed, the future change will be the value for the chloride concentration of the flood water, C_f. In the vicinity of the salt-spring areas, and for miles downstream, large salt flats are present. During each flood the salt from the flats is redissolved and adds considerably to the salt load. Elimination of the salt flats will result in a significant decrease in the chloride concentration of the flood water. Therefore, to predict the future chloride concentration of alluvial water, the value of C_f is reduced to reflect its future concentration, C_{f_1}, and the equation is rearranged and solved for a new value of C_r as follows:

$$C_{r1} = \frac{QC_q + GC_g + FC_{f1} + P - E}{R + B}. \qquad (15)$$

This new value for C_{r1} represents the future concentration of the contaminant in the alluvium at the downstream end of the one-mile stretch of river at the end of time, t, after the change has been initiated.

Also, at the end of time, t, a new value for C_q will be established at the upstream end of the one-mile stretch. This new value, C_{q1}, can be determined from a plot of the field data, C_r vs. $C_q - C_r$, as demonstrated by Figure 3. Information for this curve is from the same field data used to construct Figure 2. $C_q - C_r$ is the contaminant reduction per mile, and this difference decreases as the concentration C_r decreases. Therefore, for each new value of C_r such as $C_{r1}, C_{r2} \ldots C_{rn}$ calculated at the end of times ($t_1, t_2 \ldots t_n$) using the equation, a new value for $C_{q1}, C_{q2} \ldots C_{qn}$ can be determined by adding the appropriate value of $C_q - C_r$ from Figure 3 to C_r to find the new C_q.

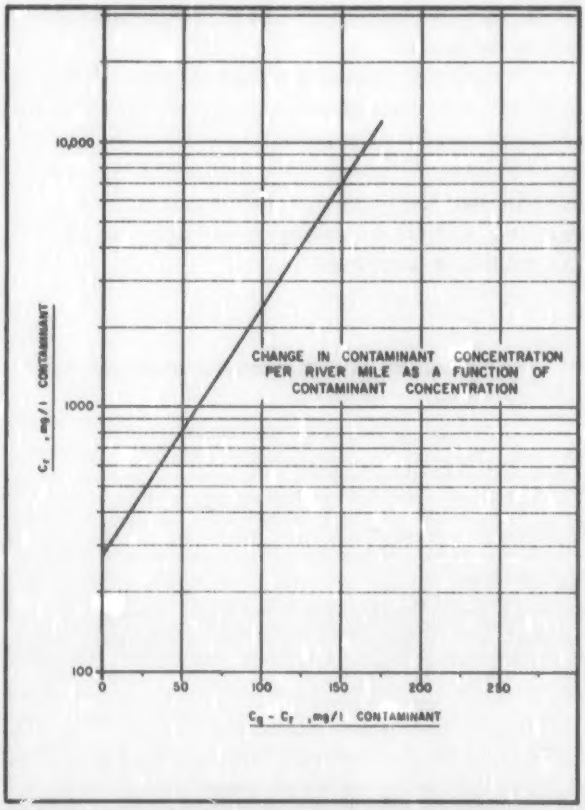

Fig. 3.An example of the rate of decay of contaminant versus contaminant concentration. Used to determine C_{qn} when C_{rn} is known.

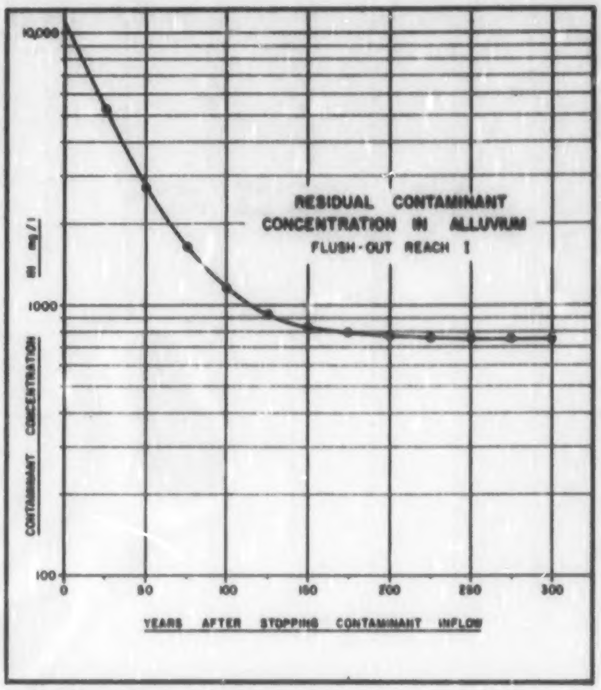

Fig. 4. An example of a future decay curve calculated from repeated iterations of the equation.

Then for the second time interval, t_2, the equation is solved as follows:

$$C_{r2} = \frac{QC_{q1} + GC_g + FC_{f1} + P - E}{R + B}. \qquad (16)$$

The future values for $C_{r1}, C_{r2} \ldots C_{rn}$ calculated by repeated runs of the equation can be plotted to show the future decay of the contaminant concentration in the alluvium and the new equilibrium level as illustrated in Figure 4.

In addition, the future contaminant load entering the surface flow from the alluvium can be determined by plotting the values of base load, BL, which is the average of BC_r over a given time period expressed in tons per year. This value may be determined using the following equation:

$$BL = \frac{1.359 \times 10^{-3}\, B\,(C_{rn} + C_{rn-1})}{2t} \qquad (17)$$

where:

BL = load from base flow in tons per year,

B = quantity of base flow over t years from equation (14),

$\frac{C_{rn} + C_{rn-1}}{2}$ = average contaminant concentration of base flow within the time interval of t_n and t_{n-1},

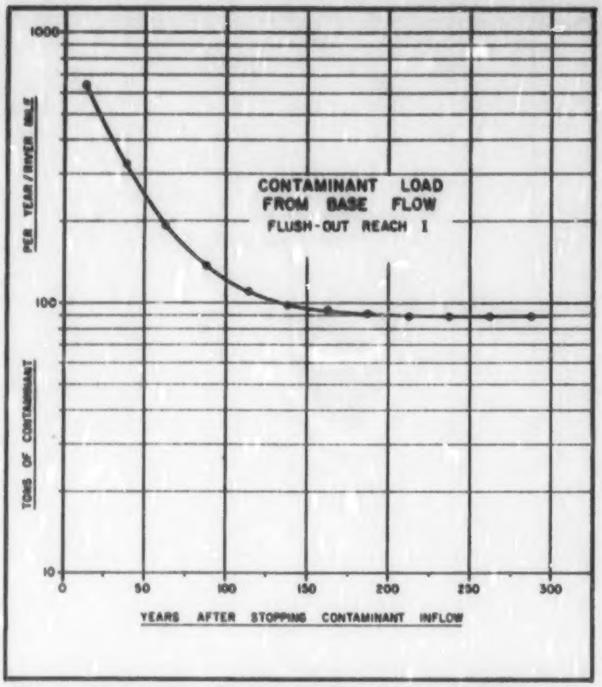

Fig. 5. An example of base load, BL, in tons of contaminant (chlorides) per year per river mile.

1.359×10^{-3} = constant to obtain tons of chloride from concentration and volume. A different constant is required for other contaminants.

Figure 5 is a plot of future contaminant load from base flow. Note that the points connected by the curve are halfway between the points on Figure 4 on the time scale because the average load between each time interval, t_n to t_{n-1}, was calculated.

If the affected river is contaminated for a long distance downstream from the source, then it should be divided into several reaches. Each factor for the equation should be calculated separately for each flushout reach. Some of the factors, such as GC_g, FC_f and even E, may change considerably between flushout reaches. The one-mile stretch used in the equation should represent the midpoint of each flushout reach. After calculating the future decay curve or residual concentration at the midpoint of each reach, new profiles of contaminant concentration vs. distance downstream from the source can be plotted at selected future times. Such a plot is demonstrated in Figure 6.

CONCLUSIONS AND RECOMMENDATIONS

The mass-balance equation herein developed has been applied to five rivers in Oklahoma and Texas. The predicted future salt flushout from the alluvium is quite different from river to river, reflecting differences in the hydrologic conditions along the rivers.

Although the equation is believed to represent a valid approximation of future flushout, the accuracy of the future predictions is dependent on the accuracy with which the various factors can be calculated. The difficulty in accurately determining the values of the various factors is fully recognized; however, the initial equation-balancing process has proved valuable in establishing reasonable values for the factors and at the same time allowing the equation to reflect observed field conditions.

It should be stressed again that the equation is presently intended for use in situations where an equilibrium exists between contaminant concentration and the hydrologic forces. It is recommended that potential users of this equation make modifications before applying the concept to contaminant plumes not yet in equilibrium with the hydrologic forces. For instance, in many such cases the value of the concentration of flood-inflow water, C_f, would not be the cause of future changes. Instead, the field-measured existing concentration gradient, $C_q - C_r$, per unit distance could be related to time, and the basic concept could still be applied. Modifications required for those applications are left to the reader.

It is further recommended that users of this mass-balance equation use a computer for the main equation and for calculating the more complicated input factors. Although the calculations may be done by hand calculator, considerable time is required both for balancing the equation and for the predictive iterations.

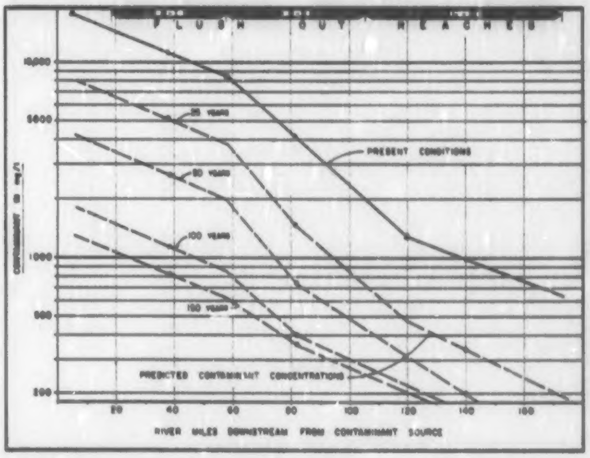

Fig. 6. An example of predicted future contaminant concentration downstream from the source based on calculated decay rates for three flushout reaches.

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DISCUSSION

The following questions were answered by John S. Fryberger after delivering his talk entitled "Quantifying the Natural Flushout of Alluvial Aquifers."

Q. *How can the equation represent a true mass-balance without concentration values for P and E?*
A. The concentration values, C, for P (precipitation) and E (evapotranspiration) are assumed to equal 1 and therefore are not shown. In a strict sense, P and E should have corresponding concentration values of C_p and C_e. From a practical viewpoint they were omitted because they are assumed to equal 1.

Also, it was pointed out during the discussion that P and E could be omitted from the main equation and used only in the water-only equation (11). This approach may be workable but has not yet been fully evaluated.

Q. *Would it be possible for chloride-contaminated alluvium (assuming no further addition of chloride) to be flushed by a single precipitation event?*
A. I cannot invision that a single precipitation event could have much effect where significant contamination is present.

Q. *Have you considered the application of thermonics equations to concentration parameters?*
A. No.

Q. *Assuming you have determined flood discharge values from a statistical analysis of discharge records, what data would you look for to aid in estimating that portion of flood discharge that enters the aquifer?*
A. The most straight-forward method would be to install water-level recorders in observation wells to measure changes in ground-water levels during flood events. However, such historical records are extremely rare and would require years to accumulate for an ongoing project. The approach used in our work is as follows: (1) From cross sections across the river, determine that water depth above river bed required to flood a significant part of the alluvial flood plain. (2) From gage-height versus flow records determine that flow that occurs at the gage height or water depth found by step 1. (3) From flood records construct partial-duration flow frequency curves at each gaging station to establish the frequency of the critical flood determined by steps 1 and 2. (4) Determine the volume of otherwise unsaturated alluvium that would be saturated by the floods determined by previous steps. (5) Multiply the flood frequency (for instance 0.2 floods per year) times t years from equation (6) times the volume of newly saturated alluvium from step 4 above. (6) For our work we have then assumed a storage coefficient of 0.2 and also that only 0.4 of the total flood inflow percolated down to mix with the normal alluvial water.

In this way we estimated a value for F independent of the F value calculated by the equations. If necessary, the least reliable independent variables, G and C_g, and lesser reliable independent variables, P and E, were adjusted so that the two methods for determining F were in reasonable agreement.

Q. *How do the precipitation and flood waters mix with the deep alluvial water, and does density stratification, play a role?*
A. Assuming that the alluvium is not highly and consistently stratified and that the ratio of vertical to horizontal permeability is relatively low, then at least some of the flood water and precipitation inflow will percolate deep, especially in rivers where the flood plain width is great compared to the depth of alluvium. Stratification due to density was most pronounced in those rivers where the alluvium was relatively deep and the flood plain narrow. In such cases both the present day and future calculated flushout rates are low.

The paper by William H. Walker was not available for publication.

Improving the Sanitary Protection of Ground Water in Severely Folded, Fractured, and Creviced Limestone[a]

by Elmer E. Jones, Jr.[b] and Charles M. Murray[c]

ABSTRACT

The construction of wells yielding safe, sanitary water in areas of severely folded, fractured, and creviced limestone depends on the retention of enough sediment in the transmission paths to restrict the movement of contaminants. The described research has been directed towards two problems—the identification of terrain and hydrologic features that indicate a lack of adequate sanitary protection of ground water in the formation, and means of improving this protection. Intermittent spring-sinks along streams or in low-lying areas indicate a reversal of ground-water flow in the area. When the water level rises above a critical level sediments that have accumulated in the formation over many years may be discharged in a few days, resulting in a loss of filtering ability.

Ground-water levels and bacteriological contamination have been monitored on four wells in Washington County, Maryland, since 1973. Ground-water quality deteriorated after tropical storm Agnes washed sediments from the formation. Techniques for preventing surface contaminant entry, improving filtering ability of the formation, and providing controlled relief of hydrostatic pressure have been studied as ways of protecting ground water in the formation.

INTRODUCTION

Many local studies have been made on bacteriological quality of individual well-water supplies. Normally, about 40% of the wells yield water of unacceptable bacteriological quality (Whitsell, 1975). In one Tennessee county, 85% of the drilled wells were contaminated with 4 or more coliform per 100 ml and 54% contained fecal coliform (USEPA, 1971). The report stated, *"Generally speaking, quality of well construction in all three counties is so poor, that one or more deficiencies threatening the safety of the source could be found in nearly every well."* Persons responsible for public health protection, ground-water development and management must consider: (1) How many wells would yield safe sanitary water if the best available technology were applied? (2) Under what circumstances is the best available technology inadequate?

The 1969 Community Water Supply Survey (USHEW, 1970) provided a partial answer to the first question. Of 621 municipal wells studied, 9% contained coliform but only 2% contained confirmed fecal coliform. This report stated, "Most of the wells having total coliform or fecal coliform densities greater than 4/100 ml were constructed, or so located as to make contamination of the water a not-too-unexpected result." Application nationally of best available technology should result in about 95% of wells yielding water of acceptable bacteriological quality.

In some areas, currently accepted best available technology is difficult to define or apply. Areas with severely folded, fractured, and creviced limestone have a higher percentage of contaminated wells than areas with sandstone or alluvial aquifers. Many limestones have low primary permeability, and ground-water movement of importance is through fractures, bedding planes, fissures, and solution channels. The sediment

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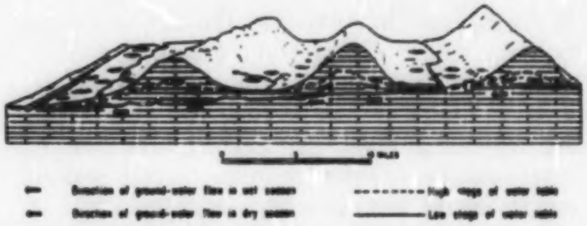
miles
Direction of ground-water flow in wet season
Direction of ground-water flow in dry season
High stage of water table
Low stage of water table

Fig. 1. Profile of a karstified carbonate terrane showing fluctuation of the water table, the uneven profile of the water table resulting from differences in permeability, and the direction of ground-water flow in wet and dry seasons (LeGrand, 1971).

contained in many of these openings provides adequate sanitary protection to the ground water entering the wells. Many of these wells, however, can be pumped at rates that will remove sediment from the formation. If enough sediment is removed, a path for transmitting contaminants from the surface to the production zone may be created. Although not commonly done, the production zone of wells constructed in severely folded, fractured, and creviced limestone can be screened and packed with sand or gravel to prevent loss of sediment during pumping.

In this paper, we will describe some hydrogeologic situations where extreme natural hydrologic events can remove large amounts of sediment from a formation and procedures that may be used to restore or improve the sanitary protection of ground water.

TECHNICAL BRIEF

Normally, water-table contours are approximately parallel with land surface contours, but they change as the water table rises and falls. Generally, water flows toward and parallel with streams and rivers. Karst aquifers are less uniform in contour because of wide differences in recharge and permeability, but the same conditions are normally anticipated. The situation illustrated on the left-hand side of the valleys in Figure 1 is considered abnormal; in dry weather, the slope of the water table is opposite the slope of the land. When the water table is high, however, ground water flows to the nearest stream. This reversal of ground-water flow has been documented by LeGrand and Stringfield (1971), but the deleterious effects on ground-water protection have not been described.

A severely folded, fractured, and creviced limestone presents an extremely complex three-dimensional flow network. The network is difficult to characterize because of the wide variability in the width, spacing, and orientation of fractures and solution channels, as well as in the size, distribution, consolidation, density and bridging of the filling sediment. But regardless of its complexity, any given part of the formation will have a path of least resistance. If the overburden is an expansive clay soil that when saturated serves as an aquiclude, the hydraulic gradient to discharge points, or openings, may be quite high as a result of artesian pressure. The openings may be old wells; spring-sinks; worm, insect larvae, or crayfish holes; or recent borings by man.

The development of a spring-sink contaminant transmission path in a karst formation initially having adequate filtration involves a series of complex events. When an opening is made through the aquiclude, discharge begins. The capacity of the upward flow of water to transport sediment is not necessarily limited by Stokes Law because the sediment is nonhomogeneous. Silt and clay in suspension increase the apparent fluid density, making it easier to suspend sand which in turn makes it easier to suspend gravel. Suspended particles too large to be transported will move to the side walls. As sediment is removed, hydraulic resistance is reduced and sediment can be transported from deeper or farther in the formation.

The discharging flow path must develop in the existing fractures, bedding planes and crevices, and must obey the laws governing sediment transport and hydraulic resistance. The mass of the largest particle that a flowing fluid can move is proportional to the sixth power of the velocity of the fluid near the particle. The hydraulic resistance of a pipe or closed channel is inversely proportional to the fifth power of the diameter or thickness of the pipe or channel. For horizontal flow sections, a channel will develop along the roof of the section and its capacity will be limited by the sediment transport capacity of the vertical segments. Thus, in a sediment-filled solution channel that is several square feet in cross section, the active pipe flow area may be less than one square inch. The flow path, to the extent permitted by the formation, will develop toward the area of highest head. An open well bore is a low resistance path through the formation that may interconnect many cracks and crevices, thus becoming a point of high head. When a well is in a discharge path, with water entering the bore at one or more points and leaving the bore through one or more paths, the water may be extremely turbid but free of bacterial indicators.

As water levels decline, discharge from spring-sinks will cease and the flow path will be able to accept recharge. How serious the hazard for ground-water contamination might be depends on the nature and availability of nearby contaminant sources, the availability of a transporting medium, and, the capacity of the contaminant transmission path. If we assume the spring discharge point was in or near a clipped pasture, runoff would include manure with fecal organisms, grass clippings, bacteria and soil particles from soil erosion and atmospheric contaminants associated with precipitation. The recharge influent could contain 10^5 to 10^9 coliform per 100 ml.

Organic debris and soil particles may lodge in the contaminant transmission path and develop a sufficient filter to reduce or eliminate bacterial contamination for a time. As the organic matter decomposes, however, the carbon dioxide produced accelerates the dissolution of adjacent limestone. Also as the water level drops, sediment in cracks and crevices may dry, shrink, and compact, once more opening the contaminant transmission paths. Thus, the gradient from any recharge point, sink or doline to the water level in a well increases. This permits increased flow velocity and sediment transport capacity. Sediment may accumulate in the well, move out into other flow paths, or be pumped from the well.

The rate of recharge and rise in the water table is important in determining the nature of the discharge from an intermittent spring-sink. A rapid rise of 10 feet or more that results in discharge will remove much more sediment from the formation than will a gradual rise over several weeks that gives clays an opportunity to expand and sediments a chance to settle before discharge begins.

Intermittent spring-sinks will not normally persist in stream bottoms. Bed load sediments having a fall velocity greater than the discharge velocity of the spring will enter the outlet. As discharge declines, progressively finer sediments can enter. When discharge ceases and recharge begins, the entire bed load contained in the recharge flow may enter. When the stream bottom is in a gaining mode, bottom areas that have been through this process may be identified by the presence of coarser bottom sediments. When the stream bed is dry, such areas may be noted by the tendency of the thin clay-silt sediment layer to dry, crack and curl over the coarser sediments which function as a capillary barrier.

When a site is identified where reversal of the ground-water gradient is removing filtering sediment and allowing contaminants to move rapidly through the formation, the ground water may need improved protection. Identifying and closing the spring-sinks normally will not produce permanent improvement; pressure will increase and discharge will probably develop at another site. A better solution is to provide an engineered path of least resistance that can discharge ground water to reduce the hydraulic gradient on the spring-sink area and prevent the entry of contaminants.

If the proper sand-water ratio and fill rate are maintained, large quantities of sand can be placed in the formation through sinks. When discharge of sediment is halted, nature will fill the flow paths. If contaminant transmission paths end in or pass through a well, a filter and screen may greatly improve water quality.

FIELD STUDY

The field study reported was conducted on farm WA10. Well WA10A was one of 24 wells being monitored for pesticides in the Hagerstown Valley, Washington County, Maryland. Farm WA10 was selected for detailed study because it appeared that pesticides might be entering Well A below the water table. Details of early field work have been published (Jones, 1973-74).

The Hagerstown Valley is part of a broad synclinal structure known as the Massanutten synclinorium. It includes the area between South Mountain on the east and Fairview Mountain on the west. The rocks in the valley are highly contorted, fractured, and contain many faults and folds.

Farm WA10 is located on the Beekmantown Group, which is made of three formations: Stonehenge limestone, Rockdale Run formation, and Pinesburg Station dolomite.

The basal formation, the Stonehenge limestone, outcrops in narrow belts over a wide area of the Hagerstown Valley. It is composed of a large lower algal limestone member, and a thin-bedded upper mechanical limestone member. The Stonehenge formation is about 750 feet thick in the Valley (Slaughter and Darling, 1961; Sando, 1957). An east-west geologic cross section of the Valley is shown in Figure 2. A partial map of the farm area is shown in Figure 3. Many small spring-sinks are around the edges of area F and along the stream.

Major operations or occurrences in the field study are listed in Table 1. Table 2 provides a brief description of the wells. Wells A and B were grouted and cased by the sinking shoe technique, using Type III A cement with a 60% water/cement

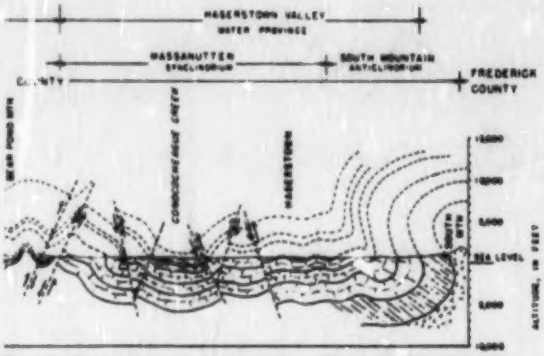

Fig. 2. East-west geologic cross section of Hagerstown Valley. Farm WA-10 is east of the Conococheague Creek area.

ratio. Well A had a contaminant entry point 79 feet below grade, which leaked air at about 2 gpm (Jones, 1973). After open bore grouting, the bore tested airtight to 100 feet at 35 psi. From initial testing, it appeared that all contaminant entry had been stopped.

Before reconstruction of Well A, chlorine demand fluctuated widely and sometimes exceeded chlorinator capacity when using standard laundry bleach. Since reconstruction, chlorine demand has not varied more than 1 ppm, the demand equivalent to a change in ammonia concentration of 0.1 ppm.

Coliform were detected in Well A on March 23, 1973 with 12/100 ml at 10 minutes pumping time. Subsequent counts were 80 on April 2, 990 on April 25, and 700 on April 27. Because nine months had passed since tropical storm Agnes occurred, it was difficult to initially associate these results with that storm.

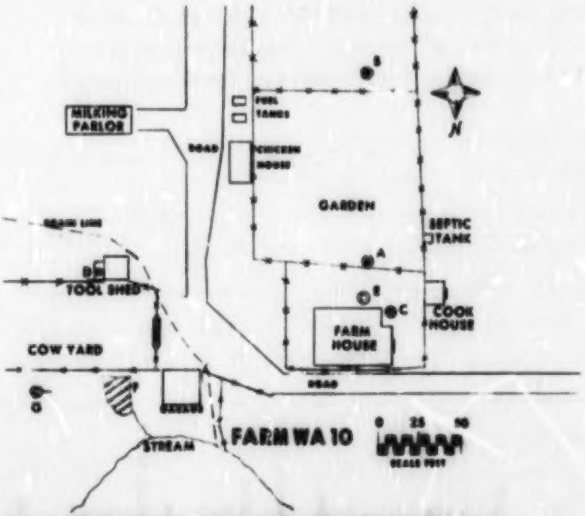

Fig. 3. Partial map of Farm WA-10 showing Wells A, original drilled well; B, new drilled well; C, dug well; D, old dug well; E, cistern; F, intermittent spring-sinks; G, old well site.

Table 1. Major Operations or Events in Field Study, 1969 – 1976

May 1969	Well B constructed, cased and grouted to 100 feet.
May 1970	Evidence of major surface infiltration noticed at Well A.
Feb. 1971	Well A reconstructed, cased and grouted to 60 feet.
Jan. 1972	Well A open bore grouted to 100 feet.
June 1972	Tropical storm Agnes, 8.1 inches rain.
May 1973	Dye test of septic tank drainage to Well A negative. Gradient from Well D to Well A observed.
July 1973	Dye test Well D to Well A.
Aug. 1973	Large sink hole filled 500 feet northeast of Well D.
Sept. 1973	Well D casing set, 2.25 yards cement grout and 10 tons sand placed.
Nov. 1973	10 tons sand placed Well D. Sand recovered from Wells A and C. Well A returned to service.
Jan. 1974	Major ground-water discharge at site known as Old Well, northeast of Well D.
Feb. 1974	Sand fill of Well D completed.
July 1974	Dye test Old Well site to Well D. Relief well installed at Old Well site.
April 1975	Well C reconstructed.
Sept. 1975	Tropical storm Eloise, 8.7 inches rain.
Dec. 1975	Relief drain line installed. All samples Well C negative for coliform.
April 1976	Well A negative for coliform, 2 consecutive weeks. First negatives since March 1973.

Water levels in the wells from June 1973 to January 1976 are shown in Figure 4. Coliform concentration for Wells A and D and weekly precipitation are shown for the same period in Figure 5. Water levels began their normal summer decline in June 1973, and coliform counts increased. That a major contaminant transmission path had developed, was obvious. Results of a dye test through the septic tank in May 1973 were negative. However because water level recordings

Table 2. Description of Farm WA10 Wells A-D

	A	B	C	D
Surface Elevation[1]	99.6	101.8	100.5	94.6
Depth, feet	131	198	32	32
Production Level, feet	127+	173	?	?
Diameter, inches	6¼	5	42	48
Specific Capacity, gpm/ft	6.3	0.07	>10?	>30

[1] Elevations are from an arbitrary datum point of 100.00 feet.

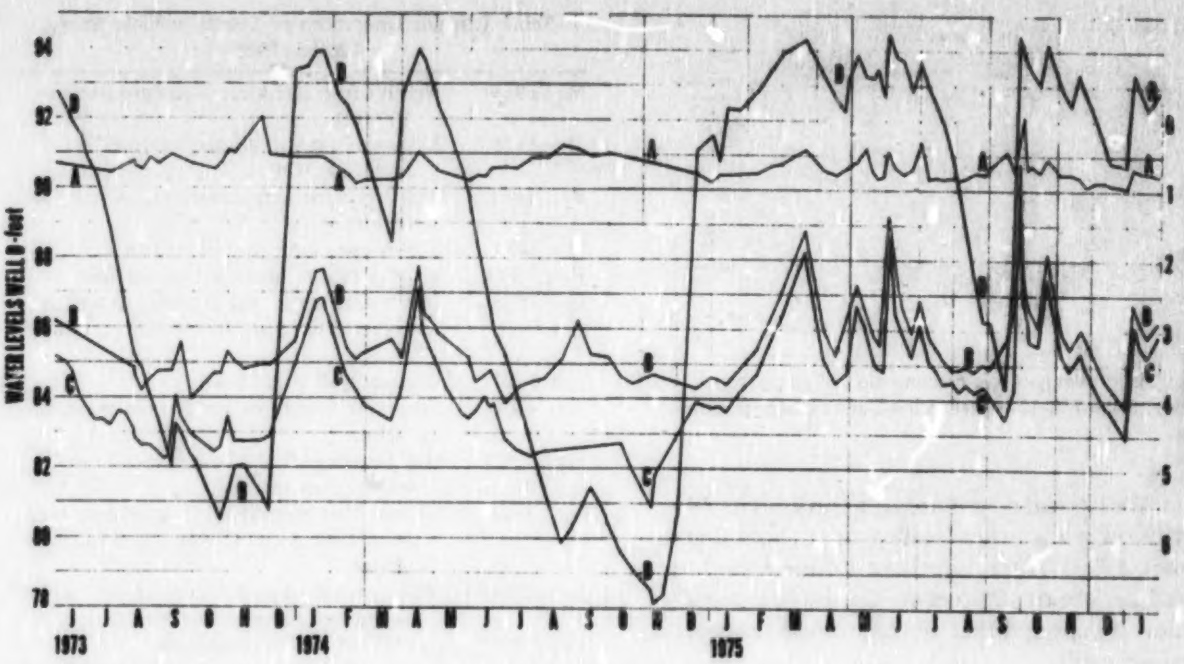

Fig. 4. Well water levels, Farm WA10, June 1973 to January 1976.

showed a gradient from Well D to Well A, a dye test was performed. The rapidity with which the dye traveled 160 feet from Well D and penetrated about 90 feet deeper to enter the bottom of Well A and the sharpness of the peak obtained (Figure 6) were surprising.

The stream north of the farmstead has a nearly level bottom from the ford northwest of the garage for 300 feet east and northeast. In July 1973, all water in the stream was entering a spring-sink hole north of the stream channel about 500 feet northeast of Well D. The sink hole was filled with 1,500 pounds of sand and 3 bags of cement, which caused water to flow down the stream and back up into the spring-sink area F. This apparently caused the coliform count to be temporarily higher in Well D than in Well A in late August.

The age and history of Well D are uncertain, but it is probably more than 100 years old. Much of the area between Well D and the spring-sink area F probably is stone fill, since many stones are removed from the fields after each plowing. The well, approximately 48 inches in diameter with stone curbing over 2 feet thick in places, is 32 feet deep and terminates in a predominantly

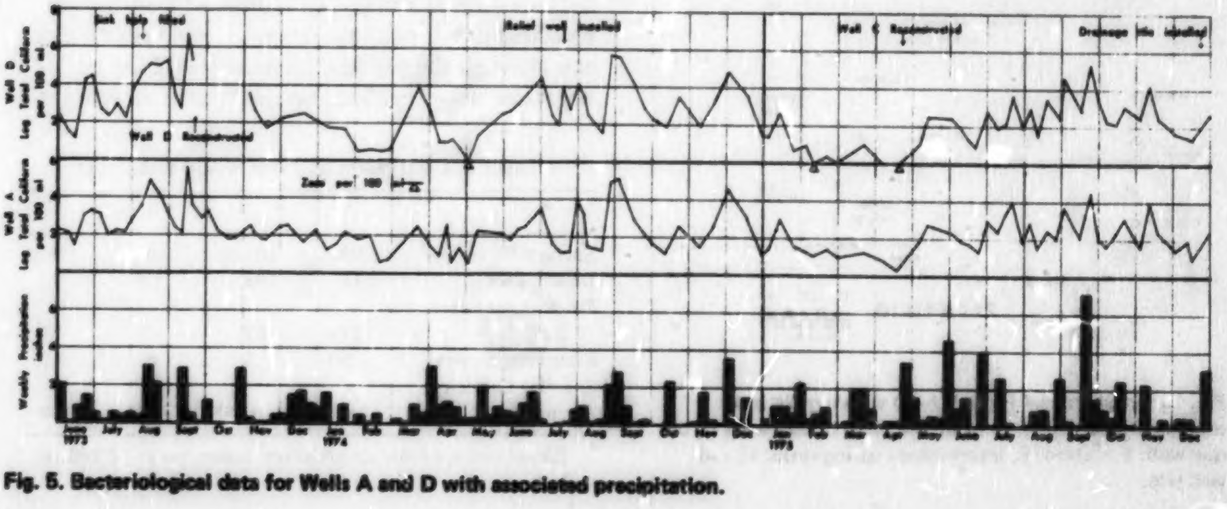

Fig. 5. Bacteriological data for Wells A and D with associated precipitation.

calcareous sediment-filled solution channel or crevice about 5 feet deep. In 1936, the well was nearly dry and a black walnut crib about 2 × 3 feet was sunk to the bottom of the sediment to obtain water.

The sediment consists primarily of fine-to-medium sand-size calcite particles. Also present are larger pieces of extremely porous (rotten) limestone, with sand and soil particles and some large organic debris. The upper few inches are predominantly organic. This organic layer may have been responsible for the wide variation in coliform counts obtained with suction samples—500/100 ml 1 foot below the water table and 5,000,000/100 ml 1 foot above the sediment.

In reconstructing Well D, we first tried to remove the bottom sediment, but sediment seemed to be entering the well as fast as it was removed. Therefore, we mixed pea gravel in the calcareous sediment by using air from a 100-cfm compressor. A slotted 5-inch casing was jetted and driven to the bottom of the sediment. A 2-inch tell tale was set alongside the casing and 2.25 yards of ready-mix grout was placed over the sediment by tremie.

After the well was producing an acceptable yield, we began to backfill the upper bore with sand dumped directly into the bore from a dump truck. About 30% of the first 10-ton load was lost into the formation. Figure 7 shows the condition of the bore 4 days after placement. A small stream of water was flowing over the sand from the north and exiting through the small hole at the south. Normally, this flow would have reached the water table through the curbing without being noticed. We washed as much sand as possible into the formation to obtain relatively stable conditions before dumping in more sand. After the second load of sand was placed in the well, sand was recovered from Wells A and C.

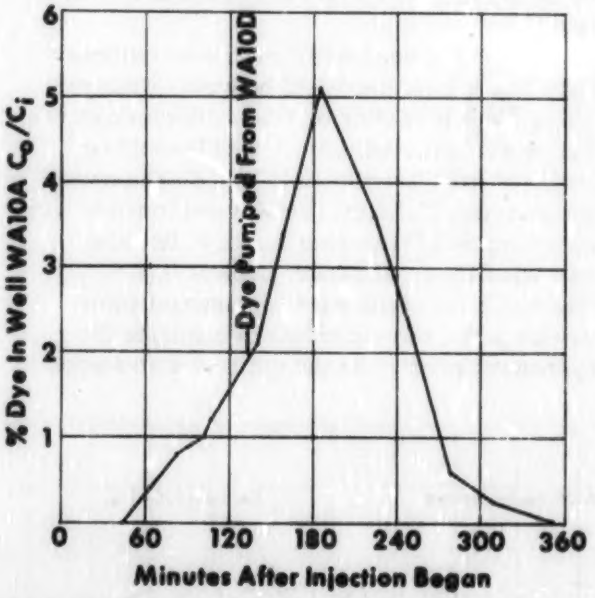

Fig. 6. Results of dye test Well D to Well A. C_i dye concentration in Well D, C_o dye concentration in water samples from Well A.

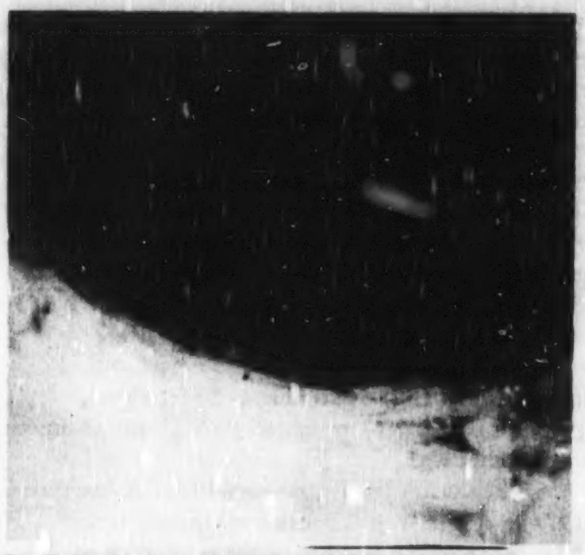

Fig. 7. Sand in bore of Well D, 4 days after placement.

Membrane filter techniques were used for the bacteriological examination of well water. For colony counts of more than 20 per 100 ml, efforts were made to obtain counts between 20 and 50 colonies with a reproducibility of ±2 colonies. In counting colonies, the sediment also was examined with a low-power microscope. Occasionally, water samples were filtered solely for sediment examination.

From the time sand was placed in Well D in November 1973 to the end of February 1974, coliform counts declined. The ground water had reversed flow and was flowing from the well sites to the spring-sink and stream area. A site about 50 feet east of the spring-sink area, known as the Old Well began discharging water, perhaps as much as 200 gallons per minute. When discharge ceased at the Old Well and area F, coliform counts soon increased in Wells A and D. Heavy rainfall in early April caused a sharp rise in water tables and a decrease in coliform counts. When water levels declined in May, coliform counts again increased. When water levels in Wells D and A were about 1 foot above stream level (91.25 feet arbitrary datum), spring-sinks were discharging and coliform counts were low. When well-water levels were lower than this, coliform counts increased after heavy precipitation, as shown in Figures 4 and 5.

Backfilling Well D with sand increased the hydraulic resistance in and near its bore, thereby increasing discharge at other points. A relief well was installed at the Old Well site G to permit ground-water discharge and prevent surface-water entry, but it did not function as desired. The 2 to 3 feet of sand and mud heaved into the casing by rapid water-table rises had very low permeability after settling. Hydrostatic heads 2 feet above ground surface were measured through the sediment in the relief well.

When water levels rose rapidly, as in April 1974, water in Well C became extremely turbid. Water entered the bottom of the bore, rose, and exited through the porous curbing, probably discharging to the stream northwest of the house. Although extremely turbid, the water was free of coliform and fecal strep organisms. The appearance of sand from Well D in both Wells A and C indicates a common flow path much of the way. It was suspected that Well C would respond rapidly to rainfall, with its water level rising sufficiently to cause water to flow to Well A. Discharge of sediment from Well C was regarded as especially undesirable because Wells A, B, and D normally had positive gradients to it.

In reconstructing Well C, 1 foot of 5-inch PVC screen with 0.040-inch slots and an open area of about 19 square inches per foot, was used with schedule 40 PVC casing. This was set in the well by hand. After setting, the casing and screen were weighted internally with 150 pounds of steel pipe to stabilize them while gravel and sand were placed. The 3,800 pounds of bagged No. 3 well gravel, 1.7 to 2.0-mm size, were placed around and above the screen. The bucket of the front-end loader used to transport sand carried about 400 pounds. After every 3 to 4 buckets of sand were placed, the fill level in the well was measured at several points. The average fill depth is shown in Figure 8. The sand hitting the water, surged the water moving sand into the formation. After the sand fill was above the water table the sand was washed and jetted into the curbing and formation. The water level was about 21 feet above the well bottom.

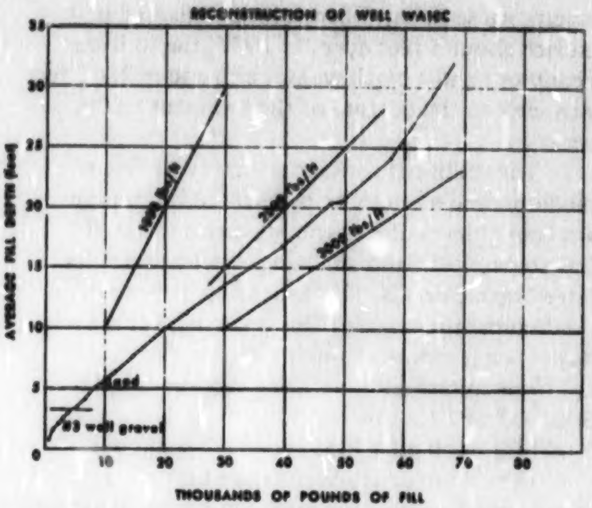

Fig. 8. Fill material use in reconstructing Well C. Fill requirements varied from 1,000 pounds per foot to over 3,000 pounds per foot.

Coliform count for Well C has been negative since December 1975, except for 2 samples having 1 coliform per 100 ml (Figure 9). Tropical storm Eloise had little effect on bacteriological quality of Well C.

In 1975, ground-water levels were extremely high. Spring-sinks fluctuated between discharge and recharge several times. When coliform counts in Well A would exceed those in Well D could be predicted from water levels in Well D. The critical elevation was 92.3 feet. This was confirmed by dye tests from Well D to a seep in area F. Because reversal of the ground-water gradient was responsible for opening the contaminant transmission paths, ways to reduce or eliminate the hydrostatic pressure on the spring-sink area were

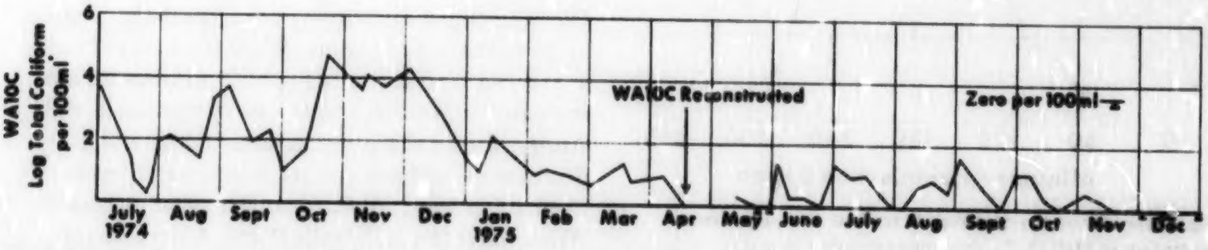

Fig. 9. Bacteriological data for WA10C for July 1974 to December 1975.

considered. The yard southeast of Well D often had standing water at high-water periods. This, in effect, was a reservoir area exerting a hydrostatic head of 1.7 to 2.7 feet of water on the spring-sink area.

In December 1975, a 200-foot-long, 4-inch drain line was installed. The first 100 feet from the creek to the fence west of the tool shed was installed in "solid" rock. In this rock, 4 to 6 inches of sand fill were placed under the fiberglass-jacketed, corrugated polyethylene tubing. In soil, the tubing was placed on 12 to 18 inches of sand fill and covered with 8 inches of sand. The sand fill provided over 400 square feet of surface area below the drain line in soil. The drain line has discharged for only a short time since installation. However, well points near Well D and the drain line indicate that the drain has reduced water levels in the sand fill at Well D.

DISCUSSION

Wells constructed in severely folded, fractured, and creviced limestone depend on sediment in cracks and crevices to protect water quality. In areas where the ground-water gradient is subject to reversal, major storms can remove in a few days the sediment accumulated over many years. Effects of this loss may not be apparent until ground-water flow returns to normal.

Differences in water level between the wells studied cannot be equated with hydraulic resistance. Differences in recharge, discharge, and hydraulic control sections may have major effects. However, when the spring-sink areas are discharging, the smaller differences in water level between Well D and Wells A, B, and C are attributed to decreased hydraulic resistance due to the removal of sediment from the formation. Water levels from December 1974 to March 1975 demonstrate this trend.

For sediment to be transported from the formation, the sixth power law of sediment transport and the fifth power law of hydraulic resistance must be satisfied. This results in the development of flow paths that have uniform characteristics, as shown by the dye test in Figure 6.

In reconstructing Well D, we made several mistakes. The well should have had a screen and sand pack. Placement of the grout over the calcareous sediments was a serious mistake. Once set, the grout could not move and prevented the sand fill from settling towards the production zone but served as a roof over unstable sediments. Use of clean concrete sand may be a mistake in some cases, because larger particles may bridge and retain finer particles that could move further into the formation.

Based on the results obtained in back-filling Wells C and D with sand, placing sand in the well in smaller quantities with a front-end loader is recommended over dumping sand directly into the well from a truck. It is believed that two factors are involved: the available energy per pound of sand to transport it through the curbing and into the formation, and the difficulty of working through 6 to 8 feet of sand. A pipe water jet could be used to settle sand in Well D more than a year after placement. Some of the difference in success in backfilling the two wells was undoubtedly due to differences in formations.

Major spring-sinks should be prevented from recharging grossly contaminated water. If it is desired to retain them as discharge points, extreme care must be taken to avoid disturbing their natural development. Several small water boils developed within 6 feet of the relief well. The burrowing by earthworms, crayfish, and other soil organisms reduces the ability of heavy soils, which function as aquicludes and are periodically subjected to positive hydrostatic heads, to adequately protect ground water from contamination.

The results obtained from Well C indicate that it is most economical to provide filtration immediately adjacent to the well. Its depth compared to Wells A and B raises questions about the trend of regulations to require deeper casing and grouting as the primary means of providing adequate sanitary protection.

SUMMARY

In the field study described, reversal of ground-water flow in an area of severely folded, fractured, and creviced limestone overlain with a heavy soil capable of serving as an aquiclude removed large amounts of sediment from the formation. The loss of filtration and reduced hydraulic resistance due to loss of sediment permitted rapid movement of surface contaminants when recharge began. Three remedial measures were investigated: improving the filtering ability of the formation, providing sanitary protection to major discharge points, and relieving the hydrostatic pressure on spring-sink areas. Best results were obtained by improving the filtration in and around Well C. The relief drain installed in December 1975 cannot yet be properly evaluated. If successful, it will reduce sediment loss from and contaminant entry into the formation.

ACKNOWLEDGMENTS

We thank the many people who assisted us with the planning and implementation of field work and testing and evaluation of results. We especially thank the farm family, without whose permission and cooperation the field studies could not have been made.

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DISCUSSION

The following questions were answered by Elmer E. Jones, Jr. after delivering his talk entitled "Improving the Sanitary Protection of Ground Water in Severely Folded, Fractured, and Creviced Limestone."

Q. *When examining for coliforms in waters having high turbidity, isn't it recommended that the multiple tube method be used instead of the membrane filtration procedure?*

A. We use the Multiple Tube MPN method when necessary and as a quality control check. As stated in the paper, we regard the examination of sediment particles when counting colonies an advantage. The dilution required for counts over 1000/100 ml greatly reduces or eliminates the effects of turbidity. For waters containing high coliform counts and microscopic organic debris, interference from nuisance organisms (Standard Plate Count) may be a greater problem than the sediment. I hope my paper will raise the question of the long-term protection of ground water when sediments interfere with the examination of well waters having low coliform counts, 0-10/100 ml.

Q. *Not knowing the geology of the area and depth to non-bearing water horizons, I'd like to ask, if possible why did you not plug the old wells, drill a new one, case and grout below the depth of the affected wells?*

A. It is hoped that our research will ultimately be of value over large areas. This research would not be practical for a single individual. Well B was constructed during the study. It also was affected by the extreme high-water levels associated with tropical storm Agnes. After April 1973, the coliform counts were less than 5% of those in Well A. At this time, I cannot recommend either a safe separation distance or depth of casing and grouting in areas where the sanitary protection of the formation is reduced by the removal of sediment from the formation by reversal of ground-water flow.

When the specific capacity of the wells are compared, Table 2, it appears that the cost of reconstructing an existing well of proven capacity may be less than the risk of drilling new wells that may be also contaminated, low yielding, or dry. In reconstructing old dug wells such as Well C and obtaining water of acceptable bacteriological quality, we are also proving techniques that can be used to close old wells to protect ground water.

In much of Maryland, 200 feet is regarded as the practical depth for obtaining wells with adequate yield. In the Hagerstown Valley, most of the wells deeper than 200 feet have specific capacities less than 0.1 gallon per minute per foot.

Land Disposal of Hazardous Wastes: An Example from Hopewell, Virginia[a]

by D. H. Walz[b] and K. T. Chestnut, Jr.[c]

ABSTRACT

In early September 1975 a Technical Committee was established by the Virginia State Department of Health to clean up and dispose of a manufacturing plant which had produced the pesticide Kepone, in Hopewell, Virginia. The Committee recommended that the hazardous wastes be buried in a virgin section of the new City of Hopewell landfill. The disposal pit was designed to encapsulate the wastes in clay and plastic. The project was completed in early March 1976. The disposal site is monitored by the Virginia State Water Control Board via an observation well and an underdrain system.

INTRODUCTION

In late July 1975 the Virginia State Department of Health was informed that a blood sample taken from an employee of a pesticide plant in Hopewell, Virginia was found to contain a significant quantity of Kepone. A subsequent visit to the plant by State officials resulted in the immediate voluntary closing of the facility. The analyses of blood from other workers and from local residents showed that the manner in which Kepone had been manufactured had created a serious health hazard for the entire community. The independent City of Hopewell, which has a population of approximately 25,000, lies at the confluence of the Appomattox and James Rivers (Figure 1).

Kepone, a chlorinated hydrocarbon, had been manufactured at this plant since early 1974. The Hopewell product then was shipped to Baltimore, Maryland where bait and/or other inert materials were added to form the marketable pesticide. Kepone was used in the United States to control ants and roaches and in Latin America as a pesticide for banana pests.

As part of an area-wide environmental and health assessment of the Kepone contamination, the Virginia State Department of Health established an

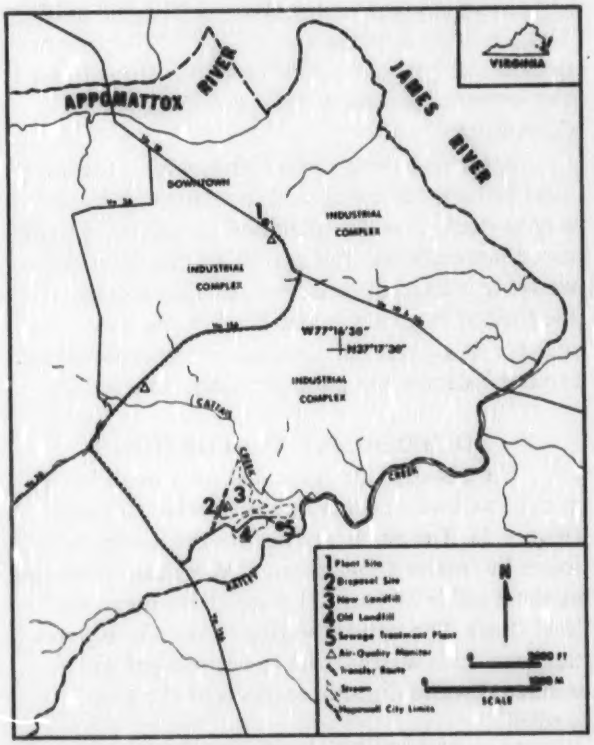

Fig. 1. Map of City of Hopewell, Virginia area.

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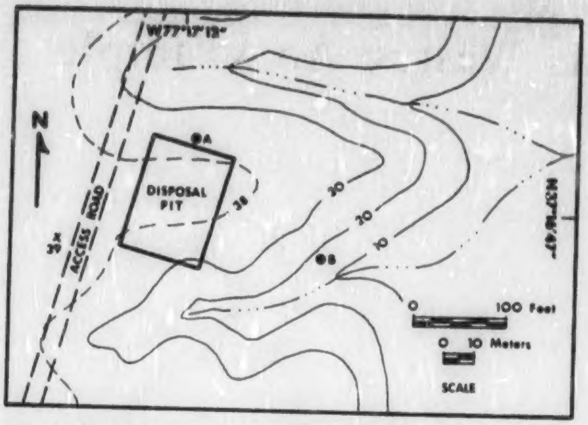

Fig. 2. Physiographic setting of disposal pit. "A" is underdrain standpipe; "B" is observation well. Elevations in feet above mean sea level.

Ad Hoc Technical Committee in early September 1975 to formulate a program for and supervise the operations of the cleanup and disposal of the Kepone manufacturing plant. The Committee was composed of the following professionals: (1) two industrial hygienists (Bureau of Industrial Hygiene, Virginia State Department of Health), (2) a groundwater geologist (Virginia State Water Control Board), (3) a landfill consultant (Bureau of Solid Waste and Vector Control, Virginia State Department of Health), (3) a civil engineer (City of Hopewell), and (5) an epidemiologist (Bureau of Epidemiology, Virginia State Department of Health). Representatives of the Virginia Air Pollution Control Board also attended several meetings of the Technical Committee.

After reviewing several alternatives, the Technical Committee concluded that the only feasible way to dispose of the plant and its related contaminated materials was to encapsulate the hazardous wastes in a lined disposal pit. A virgin section of the City of Hopewell's new landfill site was selected as a potential disposal area because of its remote location and good physiographic setting.

DISPOSAL-PIT EVALUATION

The disposal site is located on a small knoll in the southwest corner of the new landfill area (Figure 2). The site is drained on the north and south by intermittent streams. Maximum elevation of the knoll is 39 feet (11.9 m) above mean sea level (msl). The natural setting of the site isolates the hazardous wastes from the municipal and industrial waste disposal sections of the City landfill.

Three deep borings at the site indicated the presence of 28 to 30 feet (8.5 to 9.1 m) of hard clays containing laterally discontinuous lenses of clayey silts, clayey sands, and silty sands. Medium-grained sands lie beneath this upper zone of clays. These sediments are part of the Coastal Plain of Virginia.

Water was encountered in the borings at an average elevation of 5 feet (1.5 m) above msl. Observation wells drilled in the old City landfill in March 1976 suggest that the water in the borings at the disposal site probably represents a perched water table. Ground-water flow is to the southeast.

The disposal site initially was accessible from paved streets only via a power transmission line dirt road. Prior to excavation of the disposal pit, this road was widened and improved. At present, the road serves as the entrance to the new City landfill.

There are no residences within 1000 feet (305 m) of the site; the area is served by public sewer and water and is patrolled regularly.

Upon the recommendations of the Technical Committee, the Hopewell City Council donated this tract of land for the burial site in late September 1975. The State Department of Health issued a "Permit to Operate a Landfill" shortly thereafter.

DISPOSAL-PIT DESIGN

Initially, the contaminated materials which were to be landfilled consisted of the following (Figure 3):

1. a concrete pad which covered the entire outdoor working area of the plant;

2. approximately two feet (0.6 m) of soil beneath the pad;

3. process reactors, tanks, filters, and piping, chemical storage tanks, and other miscellaneous material in the pad area; and,

4. internal process apparatuses in the plant's manufacturing building.

The size of the disposal pit was determined by estimating the total amount of materials to be removed from the plant site and multiplying by a safety factor of 2 (100 percent). Estimates were received from four sources; the representative average was 2500 cubic yards (1911 m^3). When the excavation was completed, the pit measured 130 feet by 95 feet (39.6 m by 29.0 m) along the upper perimeter; 95 feet by 40 feet (29.0 m by 12.2 m) along the floor. The depth of the pit was 16 feet (4.9 m) in the center. The calculated total

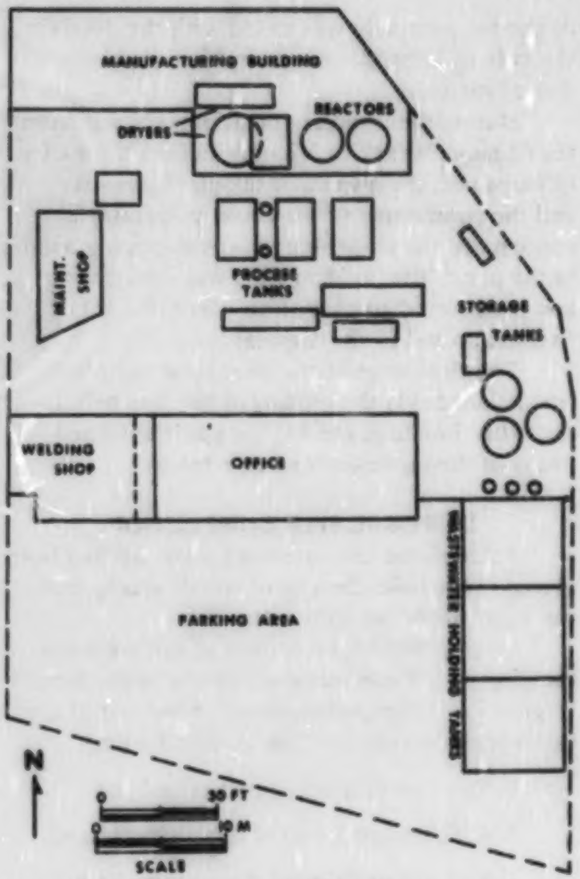

Fig. 3. Layout of Kepone manufacturing plant. Section between manufacturing building and office is outdoor working area.

available volume was approximately 4600 cu yds (3517 m³).

The Technical Committee recommended that the disposal pit be engineered with the following safeguards on the floor of the pit (Figure 4):

1. a one-foot (0.3-m) thick layer of compacted clay in the bottom of the pit;

2. an underdrain system in the top of the clay;

3. a layer of clean sand on top of the underdrain system;

4. a 30-mil (0.76-mm) thick, nylon-reinforced, plastic liner (seamed at the factory); and,

5. a protective cushion of three feet (0.9 m) and one foot (0.3 m) of clean clayey sand on the liner on the floor and sides of the pit, respectively.

The underdrain system consists of: (1) a section of 4-inch (10.2-cm) perforated plastic pipe in a 6-inch (15.2-cm) square gravel-filled trench graded to the northern end of the pit; (2) two 6-inch (15.2-cm) square gravel-filled trenches located approximately 25 feet (7.6 m) from each end of the pit and graded to the main trench; and (3) a standpipe at the northern end of the pit. If water is found in the standpipe, samples will be collected and analyzed. The water then will be removed and, if contaminants have been found in the samples, will be treated prior to final discharge.

Clean sand and clean clayey sands were imported from a nearby quarry. Sediments which were free from sharp and/or irregularly shaped objects were needed to protect the plastic liners from rupture.

The bottom plastic liner was entrenched along the top edge of the disposal pit; the top plastic liner (discussed below) was entrenched 2 feet (0.6 m) to the exterior of the bottom liner. The liners were left unsealed to allow for the escape of any gases which might be generated within the fill.

PRE-DISPOSAL CLEANUP AT PLANT

In mid-November 1975 specially trained workmen began the cleanup operation at the plant. The office building was cleaned, painted, and outfitted with showers, dressing rooms, and storage space for safety equipment needed by the workmen.

Dismantling of the plant began one week later. Under the supervision of State and private industrial hygienists, the workers dismantled the plant and discarded the scrap materials in dumpsters. When a dumpster was full, it was covered with a tarpaulin and moved to the Hopewell Sewage Treatment Plant for safe storage (Figure 1).

This phase of the cleanup was completed in early February 1976; all that remained of the plant at this date were: (1) the shells of the office and manufacturing buildings, (2) the concrete pad, (3) the wastewater holding tanks, and (4) approximately 350 55-gallon (280-l) drums which

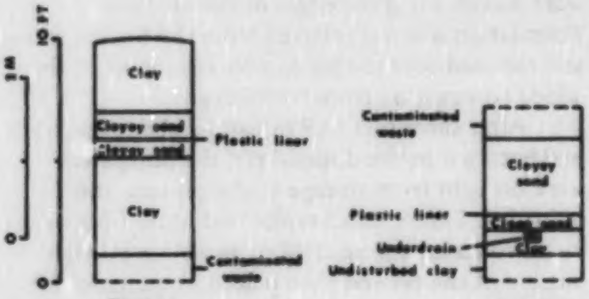

Fig. 4. Cross section through top and bottom seals of disposal pit.

contained miscellaneous Kepone-contaminated materials (these drums presently are in storage awaiting final disposal).

All rainwater and cleanup wastewaters at the plant site were drained to the holding tanks; as these tanks filled, the liquids were pumped to rail tank cars and later treated.

DISPOSAL OPERATION

On February 24, 1976, the disposal pit was ready to receive the contaminated wastes. The environmental safeguards had been placed in the pit and the liner had been covered with its clayey-sand blanket.

All personnel involved with the operation were instructed by State industrial hygienists on the use of safety equipment, which included paper or plastic suits, shoe coverings or rubber boots, safety glasses, hard hats, and respirators.

In addition to the air samplers at the plant site, Hi Vol filters were set up by the Virginia Air Pollution Control Board midway along the predetermined transfer route and at the disposal pit (Figure 1). A weather vane also was installed at the site to aid the personnel in staying upwind from the contaminated wastes.

At the plant site the concrete pad which served as the foundation for the outdoor working area was ripped up. Materials were transferred to the plastic-lined beds of tandem dump trucks and covered with tarpaulins. At the disposal site the loads were dumped along the edges of the pit; the trucks were swept out and returned to the plant site.

The equipment at the disposal site included a small bulldozer and a crane with a 1-cu yd (0.76-m^3) clamshell bucket. The first Kepone-contaminated materials placed in the pit consisted of sections of the concrete pad and the soil from underneath the pad. Concrete slabs were laid flat in the center of the pit; concrete footings and curbings were placed along the edges of the pit floor. Foundation soil was released from the bucket when still elevated over the pit to take advantage of the added compaction from freefalling.

After three feet (0.9 m) of the above materials had been put in the disposal pit, the dumpsters were brought from storage to the pit area and unloaded. Filters, small tanks, and other fiber or metal containers were crushed as well as possible outside of the pit and then placed in the hole. All pipes, worm screws, angle iron, aluminum, copper, and iron sheets were placed horizontally in the pit. Clay, which was stockpiled during the excavation of the pit, routinely was mixed with the debris in the hole to fill voids created by the irregularly shaped materials.

During the disposal operation, officials from the Commonwealth of Virginia and from the City of Hopewell, the owners of the plant property, and the contractors of the cleanup operations condemned the cinder-block manufacturing building at the plant. The building then was demolished and transported along with two feet (0.6 m) of foundation soil to the disposal site.

The final stage of the cleanup at the plant site included: (1) the grading of the area behind the office building, and (2) the puncturing and filling of the wastewater storage tanks.

DISPOSAL-SITE COMPLETION

After all the contaminated materials had been placed in the hole, the pile of rubble was spread to the edges of the pit with a drag line.

To prevent the infiltration of surface water, the disposal pit was sealed across the upper surface (Figure 4). The impermeable cap consisted of the following materials (in order of installation):

1. four feet (1.2 m) of compacted clay;
2. 8 inches (20.3 cm) of clean clayey sand;
3. a 15-mil (0.38-mm) thick plastic liner;
4. one foot (0.3 m) of clean clayey sand; and,
5. three feet (0.9 m) of compacted clay.

The site was graded and sloped to prevent the ponding of surface water and seeded to prevent erosion. Signs were erected at the four corners of the pit to warn against trespassing and excavation in the area; a bronze plaque affixed to a granite slab was installed as a permanent identifying marker at the center of the west face of the pit; and a statement which identifies the contents of the disposal pit was attached to the property deed. The disposal project was completed in early March 1976.

The disposal pit was uncovered from the beginning of the disposal operations on February 24, 1976 until the top plastic liner was installed on March 6, 1976. During this period, the weather conditions were excellent; the average maximum temperature was 76° F (24.5° C); minimum averaged 48° F (8.9° C). Thirteen-hundredths of an inch (0.33 cm) of rain fell the night before the plastic cover was installed; the top clay seal, however, already had been emplaced at this time. The hazardous-waste materials, therefore, were essentially dry when landfilled.

MONITORING PROGRAM

In September 1975 a 28-foot (8.5-m) deep observation well was installed downslope from the site (Figure 2). A number of background water-quality samples were collected from the well between October 1975 and January 1976. The underdrain and the observation well are part of an area-wide monitoring program by the Virginia State Water Control Board. During the initial phase of this program, the collection of water samples from the well and the inspection of the underdrain have been made on a monthly basis.

SUMMARY

1. The manner in which the pesticide Kepone had been manufactured had created a serious health hazard for the Hopewell, Virginia area.

2. An Ad Hoc Technical Committee, which was established by the Virginia State Department of Health, concluded that the only feasible way to rid the community of the source of Kepone was to bury the plant and its contaminated contents.

3. A disposal pit at the new City of Hopewell landfill site was designed to contain the wastes in an environmentally safe manner.

4. The hazardous wastes were landfilled from February 24, 1976 to March 6, 1976. During this period, weather conditions were optimal for this type of project.

5. The disposal site presently is monitored on a monthly basis by the Virginia State Water Control Board via an observation well and underdrain standpipe.

ACKNOWLEDGEMENTS

The authors wish to extend their appreciation to the other members of the Technical Committee: C. R. Fortney and J. A. Saunders, Industrial Hygienists, F. E. Hughes, Jr., P.E., and R. S. Jackson, M.D.

DISCUSSION

The following questions were answered by D. H. Walz after delivering his talk entitled "Land Disposal of Hazardous Wastes: An Example from Hopewell, Virginia."

Q. *Who carried out the disposal operation and who paid for it?*
A. The cleanup and disposal operation was performed by private industry at an estimated cost of $350,000 under the supervision of officials from the Commonwealth of Virginia.

Q. *Do you expect the material to degrade in the course of time? If not, is this the only feasible way of disposing of the material without creating environmental hazards?*
A. The greatest problem in dealing with Kepone has been the lack of information available on the response of Kepone to the environment. We do not know if Kepone will degrade with time. The major desire of the Committee was to rid the community of this environmental hazard as quickly as possible; the Committee investigated several alternatives but found encapsulation to be the most viable in terms of time and money.

Q. *What type of plastic in the liner? Will organics used in Kepone manufacture react with the plastic liner?*
A. The plastic used was polyvinylchloride (PVC). Prior to approving the use of PVC, the Committee consulted with EPA (Washington) and a California firm which was preparing a report on liners for EPA. Both sources stated that PVC was an acceptable material for use with Kepone.

Q. *What about the cement or bonding agent used to join the panels of the liner; will the seams be a potential zone of weakness?*
A. The liner has a guarantee of 20 years; realistically, the Committee felt that the liner would serve as a short-term barrier until the materials in the fill stabilized.

Q. *What provisions were made to monitor the quality of any gases escaping the site?*
A. During the disposal operation, hexachlorocyclopentadiene (HCP), a raw material in the manufacture of Kepone, did volatilize. There is evidence from samples collected at the plant site that HCP bonds to particles because it is an unsaturated compound and, therefore, is subject to degradation.

A Hi Vol air sampler was in continuous operation at the disposal site during the operation. There are no plans, however, to sample the air quality at the site in the future unless the situation warrants it.

Q. *Is there a risk of spontaneous combustion within the landfill?*
A. No, the landfill is a heat sink and does not contain a large amount of oxidizable materials.

Q. *Was the possibility of reacting the insecticide with some other chemical compound, one that would result in a less hazardous product, considered?*
A. Yes, but the Committee could not find such a chemical compound within the time constraints of the project.

Q. *What effects does Kepone have on people who came in contact with it?*
A. The acute chronic effects on *people* consist of neurological problems (tremors, aberrations of eye movements, anxiety, and nervousness), sterility, and changes in liver functions. In tests conducted by the National Cancer

Institute, there was evidence of cancer formation in laboratory *animals.*

Q. ***The "60 Minutes" presentation mentioned that concentrations of Kepone were discharged into the James River. What monitoring has been done in the ground-water areas near the river and what are the results?***
A. In March 1976 the Commonwealth of Virginia installed ten observation wells throughout the Hopewell area. In addition numerous domestic- and public-supply water wells were sampled in critical areas. Kepone has been found in wells on the north side of the plant site but only in concentrations at or very near (the highest has been 0.08 ppb [μg/l]) the detectable limit (0.02 ppb [μg/l]) for Kepone. Samples from wells near the Sewage Treatment Plant show levels of Kepone at or below the detectable limit; samples from wells along the river show no detectable Kepone.

Analyses of soil and core samples show that the bulk of Kepone has not migrated below the top six inches (15.2 cm) of soil. It must be noted, however, that these analyses represent short-term results from a potentially long-term reaction with the environment.

Q. ***What about rodent control?***
A. Officials from the State Department of Health routinely check the disposal site. If problems do arise from the burrowing of animals, growth of seedlings, trespassing, or other unpredictable events, they will be handled immediately.

Q. ***Has consideration been given to installing a second observation well in the drainage north of the present one?***
A. Ground-water flow in the area is to the southeast; any possible contamination would show up in the present well first. Other wells could be constructed if necessary.

Q. ***How long will it take liquids to migrate through the clay barrier?***
A. No permeability tests were made on the clays; field evidence indicated that such tests were not necessary.

Land and Water Use Impacts on Ground-Water Quality in Las Vegas Valley[a]

by Robert F. Kaufmann[b]

ABSTRACT

Marked changes in the occurrence and quality of near-surface ground water in Las Vegas Valley, Nevada result from urban and industrial land and water use practices. In-valley recharge has increased tenfold in the period 1943 to 1973 and now amounts to about 40,000 acre-feet/year (49.3 million m^3). Ground-water flows leaving the Valley have increased from 250 acre-feet/year to about 12,000 acre-feet/year.

Twenty to 400 tritium units (T.U.) in shallow ground water confirm widespread addition of recent recharge. Trend-surface analysis of recent water-quality data for depth intervals or "slices" of 0 to 50, 51 to 100, and 101 to 300 feet (0 to 15.2 m, 15.5 to 39.5 m, 30.8 to 91.4 m) revealed that natural trends below a depth of 50 feet are explainable in terms of broad hydrogeologic conditions. From 0 to 50 feet quality is highly irregular and markedly more influenced by land and water use practices and waste disposal in particular. Chloride, TDS, and nitrate are particularly diagnostic of return flows as is spring development and (or) a rising water table resulting from increased recharge and low vertical permeability. Statistical tests on water-quality data for the period 1912 to 1968 yielded generally insignificant change with time. However, the extreme paucity of the data base makes any conclusion questionable.

More efficient irrigation practices could reduce the present irrigation water demand by 15,000 acre-feet/year and reduce return flows by 11,000 acre-feet/year. Return flows by the year 2000 could easily amount to 75,000 acre-feet/year or about three times the total water budget of the Valley prior to urbanization. Therefore, ground-water problems are likely to worsen and, if present monitoring practices prevail, go unnoticed.

INTRODUCTION

The study was designed to determine the effects of past water use on ground-water quality in Las Vegas Valley and provide a basis for projecting both physical and hydrochemical changes which are likely to result from increasing water utilization in the area.

Las Vegas Valley is an intermontane arid basin containing a thick sequence of alluvial sediments arranged vertically and laterally in a complex system of aquifers and aquitards. The most permeable sediments are in the northwestern and west-central parts of the Valley where the principal volume of ground-water extraction has occurred to date, primarily from confined aquifers at depths of 250 to 1,000 feet (see Figure 1). Except in the west-central area, these aquifers are overlain by extensive layers of poorly permeable clay, silt, or caliche. Separating these productive zones from generally less permeable sediments to the east is a series of faults and associated scarps which impede eastward ground-water flow. The eastern part of the

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[a] Presented at The Third National Ground Water Quality Symposium, Las Vegas, Nevada, September 15-17, 1976.

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Valley is characterized by primarily fine-grained sediments and a shallow water table.

Under natural conditions, recharge to the Valley fill was a result of precipitation in the surrounding mountains, primarily those to the west and north. Flow from the recharge areas followed an easterly or southeasterly path which subsequently involved lateral and then upward movement in the valley fill either as diffuse seepage or as localized flow towards springs discharging along fault planes. Both the seepage and reinfiltration of the spring discharges resulted in recharge of the near-surface zone, discharge from which was by evaporation and, more importantly, evapotranspiration. There was no pronounced flow of ground water from the Valley and the principal discharge areas such as the Meadows (Las Vegas) were well removed from the present locus of discharge, Las Vegas Wash.

FACTORS AFFECTING RETURN FLOWS

The principal sources of recharge to the near-surface system include irrigation return flows, septic tank and sewage treatment plant effluents, industrial effluent ditches and disposal ponds, and upward leakage from deeper, artesian aquifers. Discharge from the system occurs as direct evaporation, evapotranspiration from phreatophytes, and discharge to surface-water courses and to Las Vegas Wash in particular (see Figure 1). Under natural conditions prevailing until the early 1940's, the only ground-water outflow from the Valley was about 250 acre-feet/year of underflow in Las Vegas Wash. By 1972 an additional 12,300 acre-feet/year of underflow surfaced in the lower reaches of Las Vegas Wash (Kaufmann, 1971; Westphal and Nork, 1972). Extensive mesquite groves and stands of saltbush that formerly discharged 25,000 acre-feet/year have largely been removed and replaced with suburban sprawl and ubiquitous green lawn. Exclusive of Henderson and the BMI effluents, recharge to the shallow aquifer in 1973 amounted to 38,500 acre-feet/year (Patt, 1976) and represented water which was (1) going into storage, (2) leaking downward to recharge the deeper aquifers, and (3) flowing laterally toward Las Vegas Wash.

GROUND-WATER QUALITY

Despite the intense urban/suburban development and resulting water use in the Valley, past documentation of ground-water quality, regardless of depth, is extremely deficient. Limited discussions of water quality are presented in Mendenhall (1909), Carpenter (1915), Hardman and Miller (1934), Maxey and Jameson (1948), and Malmberg (1965). The only water-quality map produced prior to the present study depicted zones or regions of TDS as indicated by specific conductance. Although an extensive water sampling program of the District Health Department resulted in several thousand water analyses for the period 1968 to 1973, there was no attempt to synthesize and interpret the data. Municipal wells of the Las Vegas Valley Water District and the City of North Las Vegas were first monitored on a regular basis in 1969.

Trend-Surface Analysis of Water Quality

The trend-surface technique involves fitting polynomial surfaces to map data by means of a general linear model incorporating a least-squares fit of a planar or curvilinear surface to the observed data. In the past, trend-surface analysis has been applied primarily to stratigraphic, structural and sedimentation problems but the method has not been adequately tested and applied to ground-water quality studies, particularly those in which there is a need to reduce and generalize a great mass of chemical data, some of which are of questionable veracity. It was reasoned that broad over-all trends are analytically more useful in describing variations present in Las Vegas Valley.

Available chemical analyses for ground water were screened for depth of well, year sampled, and accuracy of data. Analyses from January 1, 1968 to the present were selected for depth intervals 0 to 50 feet, 51 to 100 feet, and 101 to 300 feet providing the anion:cation ratio was in the range 0.9 to 1.1.

First order or regional trends in TDS for the three depth intervals considered are shown in Figure 2. At depths of 101 to 300 feet, the progressive increase in mineralization along the valley-wide flow path is primarily a function of chemical quality of water in the recharge zone; the type, distribution and absorptive capacity of the geologic matrix; the porosity and permeability of the rocks and sediments; and the course of water along the flow path. Differences in absolute concentrations and concentration gradients for the three depth intervals are a result of natural and man-induced factors.

The role of natural hydrogeologic controls on ground-water quality is shown in more detail in Figure 3 which is a hand-contoured map of TDS at depths of 200 to 300 feet. In general, the pattern for TDS for the zone from 101 to 300 feet shows progressive increase along the flow path.

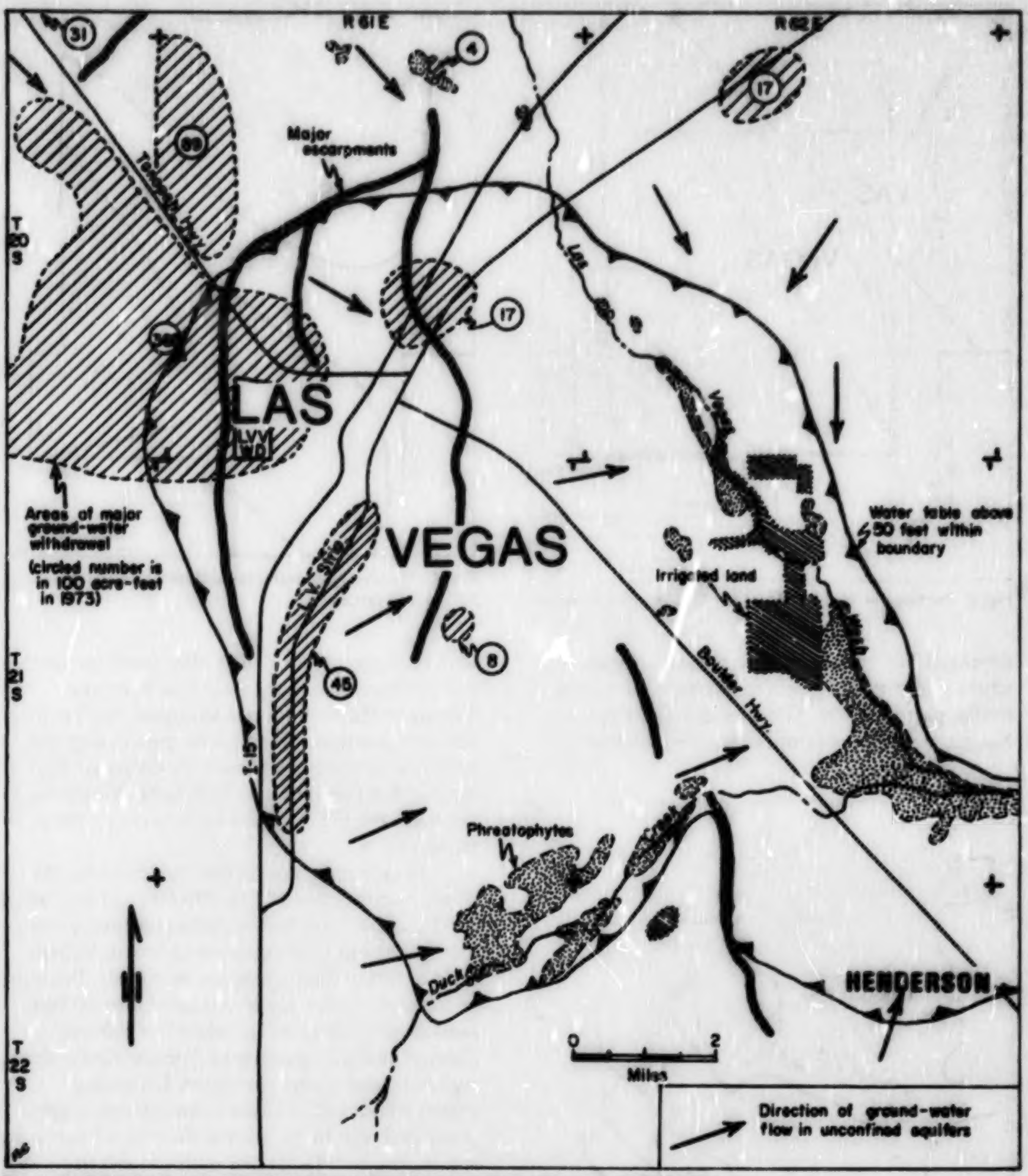
31
R61E
4
R62E
17
89
Tonopah Hwy
Major escarpments
T 20 S
36
17
LAS
LVVWD
Areas of major ground-water withdrawal
(circled number is in 100 acre-feet in 1973)
VEGAS
L.V. Strip
45
8
T 21 S
I-15
Las Vegas Wash
Irrigated land
Water table above 50 feet within boundary
Boulder Hwy
Phreatophytes
Duck Creek
HENDERSON
T 22 S
N
0
2
Miles
Direction of ground-water flow in unconfined aquifers

Fig. 1. Hydrogeologic and cultural features influencing ground water in Las Vegas Valley.

This is expected because the dominant sink is the lowland area centered on Las Vegas Wash, the point toward which all ground water flows. The mineral content of deep ground water in the southeasterly portion of T21S/R62E rapidly increases from 1,000 to 3,000 mg/ℓ in a distance of about four miles. Comparing this with the change in the distance from the prime recharge area in the Spring Mountains to the 1,000 mg/ℓ contour line, (roughly 30 miles or 48.3 kilometers), gives an indication of the role played by sediment composition and permeability of fine-grained valley fill.

Sulfate concentrations in ground water at

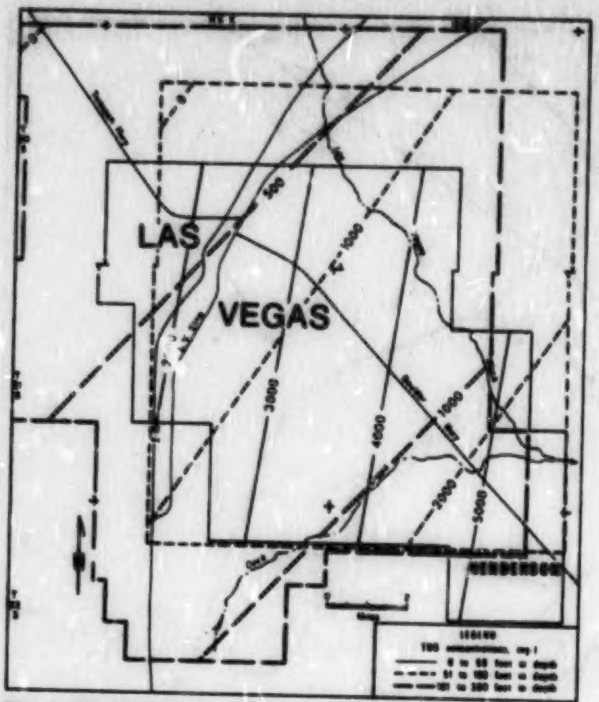

Fig. 2. First-degree trend surfaces for TDS in ground water.

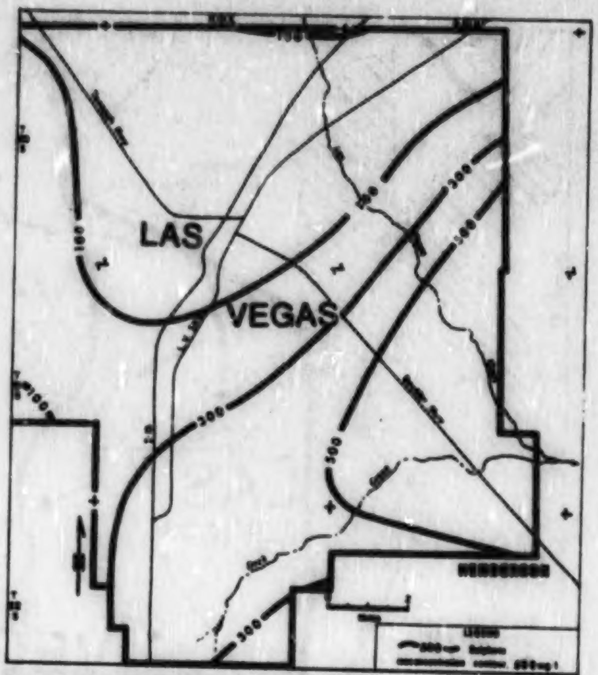

Fig. 4. Fourth-degree trend for sulfate in ground water 101 to 300 feet deep.

depths of 101 to 300 feet are shown in Figure 4 which is a fourth-degree trend surface showing a similar pattern to the TDS plot discussed above. Minimum sulfate concentrations are associated with recharge entering the Valley from the north and northwest, with minor inflow from the northwest flank of Sunrise Mountain. Inflow from the west, southwest, and south, particularly the latter, is enriched in sulfate with the result that most of the ground water in Paradise Valley does not meet the U.S. Public Health Service (1962) standard.

Nitrate and chloride trend surfaces for the depth interval from 101 to 300 feet had the lowest coefficients of correlation, indicating numerous local variations in comparison to broad, Valley-wide trends in the parameters previously discussed. In the case of chloride at depths of 0 to 50 feet, most of the variations are related to effluent disposal and are superimposed on the first-order, regional trends shown in Figure 5. The low concentrations of chloride below 50 feet suggest prime recharge to the alluvial fill comes from source area(s) low in chloride and is characterized by short residence time, or relatively short flow paths, or both. This suggests recharge to the valley fill may also be associated with movement in the carbonate aquifers rather than only in the alluvial aprons flanking the carbonates.

Rapid urban and suburban growth in Las Vegas Valley for over 30 years has involved consistent disruption of natural drainage paths and associated salt flux processes of the native soils. A typical

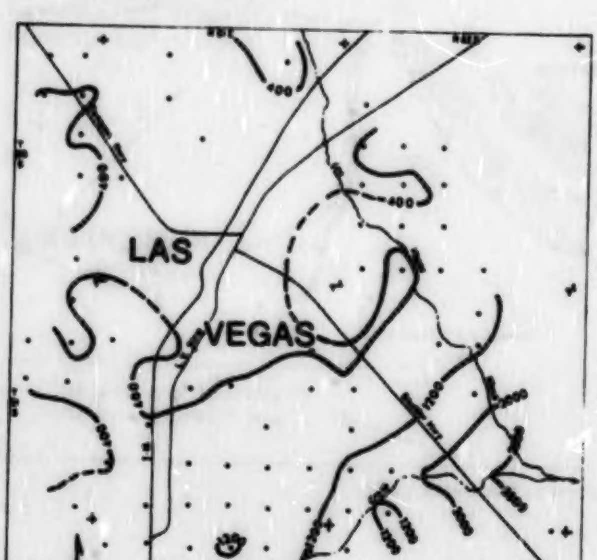

Fig. 3. Concentrations of TDS in ground water 200 to 300 feet deep.

leveling operation for a subdivision in the central part of the Valley is shown in Figure 6. For overland and open-channel flow, the net effect has been to collect and channelize flow through urbanized areas, impede or block sheet flow, and to reduce the discharge capacity of lesser tributaries. This had led to localization of runoff and increased local recharge to the near-surface zone. Similarly, removal of caliche layers and hardpans combined with the introduction of wastewater return flows from interfluvial areas, formerly characterized by sheet runoff and little if any natural recharge, provides greater opportunity for increased infiltration and increased salinity in shallow ground water.

Fig. 6. Deep soil excavation and land leveling associated with subdivision construction.

Effects of Return Flows on Water Quality

Nitrate, chloride and TDS concentrations are particularly diagnostic of return flows beneath the (1) principal urbanized portions of the Valley, (2) areas of sewage disposal, (3) areas irrigated with sewage, and (4) the industrial area in Henderson. The principal sources of nitrate contamination are shown in Figure 7. Detailed documentation of water use and return flows presented in the study by Kaufmann (1976) and by Malmberg (1965) reveal returns from sewage effluent, industrial wastes, cooling water, and septic tank systems have infiltrated the near-surface aquifer since 1912. In 1955 alone, 7,000 acre-feet of effluent infiltrated in the eastern part of the Valley, whereas, total annual recharge to the near-surface aquifer over the entire Valley was estimated at 14,000 acre-feet (Malmberg, 1965). For 1973, Patt (1976) estimated about 18,000 acre-feet. At the present time, there are 5,591 irrigated acres in the Valley which receive 47,000 acre-feet/year, only 42 percent of which is consumptively used. Clearly, recharge stresses on the shallow system are pronounced and warrant analysis of their effects on water quality.

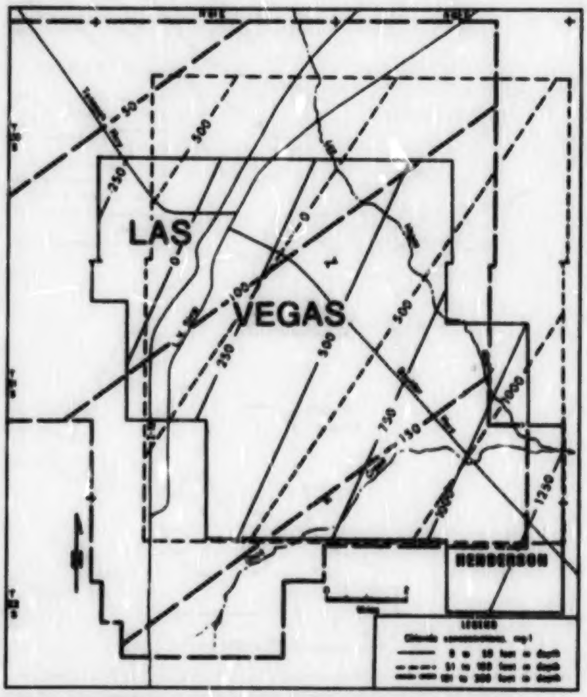

Fig. 5. First-degree trends for chloride in ground water.

Sewage effluent applied to the Paradise Valley and Winterwood golf courses since 1960 and 1965, respectively, has caused little change in TDS, but nitrate in shallow ground water increased to as much as 140 mg/ℓ. Phosphate in one well was 8.8 mg/ℓ or approximately 88 times the normal background concentration. Sewage effluent from individual treatment plants associated with several major hotels and individual septic tanks also contributed to nitrate concentrations in ground waters. By 1973 approximately 4,000 septic tank installations in the Valley, principally distributed as shown in Figure 7, contributed an estimated 1,750 acre-feet/year of wastes to the near-surface aquifer (Patt, 1976). Other miscellaneous nitrate sources include the LDS Church Farm and adjacent farms, and the sewage treatment plant and sewage-irrigated golf course at Nellis Air Force Base.

The trend surfaces for nitrate are revealing, albeit in a negative fashion. Whereas, nitrate in the zone from 0 to 50 feet is regularly distributed and quite high in the eastern part of the Valley, at depths of 101 to 300 feet it is irregularly distributed and generally quite low in concentration. Therefore, waste disposal, largely in the eastern part of the

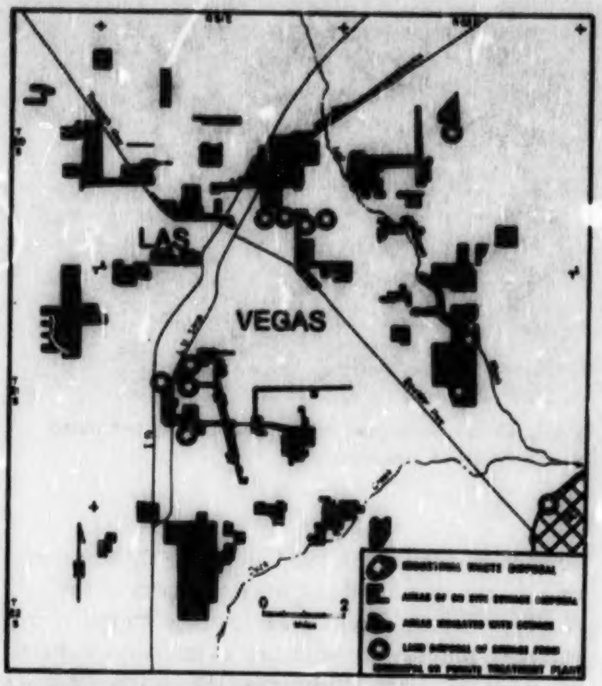

Fig. 7. Sources of nitrate in ground water.

Valley, dominates the shallow nitrate pattern. Although the volume of return flow from such nonpoint sources is relatively small, the water-quality effects in terms of elevated levels of nitrate are readily noticeable.

Temporal Changes in Water Quality

Ground-water quality data from previous studies, supplemented with analyses from the present study, were analyzed for temporal changes. Excluding (1) recent (1969-present) analyses from producing wells in the Las Vegas Valley Water District and North Las Vegas well fields, (2) the water-quality data collected by the Health Department or (3) the present study, there are approximately 412 analyses extending over an area of 150 square miles. These were collected from wells ranging from 8 to 1,700 feet deep, although 44 percent of the historical water-quality data prior to 1968 pertain to wells of unspecified or unknown depth, thereby reducing their scientific value. Furthermore, only two percent of the available analyses document very shallow water quality in the depth interval from 0 to 100 feet, compared to 22 percent from wells deeper than 400 feet. The remaining 32 percent of the wells range in depth from 101 to 400 feet. In township 21/62 and 22/62, deteriorating ground-water quality from return flows and solubilization of evaporites from a rising water table is difficult to document because there are essentially no chemical data for the unconfined aquifer. Similarly, townships 20/60, 21/60, and 22/60 are devoid of baseline water-quality data; hence changes in the system, from a multitude of causes, are and will continue to be extremely difficult to document. This is unfortunate, considering the rapid urbanization underway in the eastern portions of the Valley. Although the data for townships 20/61, 21/61, and 22/61 extend over a period of approximately 45 years, the paucity of analyses makes it difficult to define the chemical state of the system at given points or certain periods. With the exception of townships 20/61, 21/61, and 21/62, essentially nothing is known about historical ground-water quality prior to the period from 1955 to 1962, despite the fact that since the early 1940's sewage and industrial wastes were deliberately allowed to infiltrate in townships 20/62, 21/62, and 22/62.

Wells in various parts of the Valley for which historical chemical data were available were grouped as shown in Figure 8. The groups were selected using criteria of (1) generally similar hydrogeologic and water-quality conditions within each group, (2) sufficient number of wells to constitute a minimal sample size for the statistical tests, and (3) different time periods in which

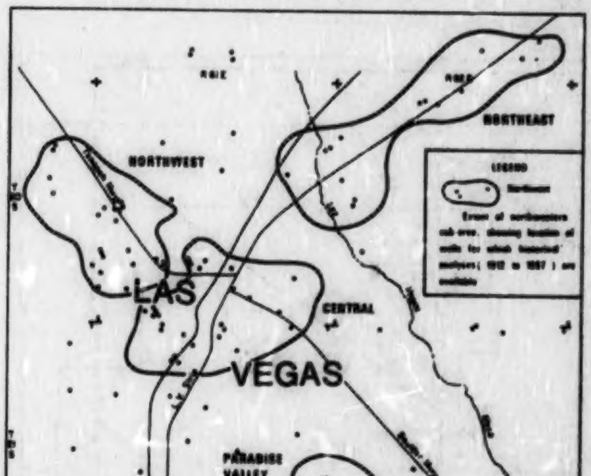

Fig. 8. Locations of wells sampled between 1915 and 1972.

changes in water quality could be compared. Within each areal grouping of historical ground-water quality data, the values for TDS and chloride were analyzed for change through time by means of two nonparametric tests, the Mann-Whitney test and, in cases where sufficient data were present, the Kruskal-Wallis test for one-way analysis of variance (Siegel, 1956). The test results indicated that from 1912 to 1972 essentially no significant change in ground-water quality occurred anywhere in the Valley, regardless of the individual time periods considered.

Return Flows of Industrial Origin

In terms of both flow volume and chemical concentration, the major industrial waste disposal operation adversely affecting ground-water quality is the industrial complex in Henderson. Now known as Basic Management, Inc. (BMI), the complex began operations in the early 1940's and initially produced magnesium metal. Starting in 1945 to 1950, various chemical manufacturing and minerals' processing industries have been operating the complex. Throughout the periods of operation, liquid effluents have been conveyed via open ditches to tailings ponds where disposal has been by evaporation and percolation.

The study involved initial assessment of lithologic boundary conditions and water quality (Kaufmann, 1971), followed by a combined paper analog and digital analysis of the flow regime (Westphal and Nork, 1972). Ground-water inflow was first estimated by Kaufmann (1971) for calendar year 1970 by taking into account the total monthly surface-water flows in the Wash at (1) the sewage treatment plants, (2) the midpoint adjacent to the tailing ponds and (3) at the outlet. Net average daily ground-water return flow in the upper reach of the Wash is 10.3 acre-feet, much of which exits the tailing ponds. Lesser amounts are from commercial irrigation with sewage, and "other" general return flow from sources such as lawn watering or golf course irrigation. Daily accretion in the reach below Pabco Road is approximately 17.4 acre-feet for a total of 27.7 acre-feet. In contrast, ground-water returns prior to extensive in-valley water use were about 0.75 acre-feet per day.

Variations in ground-water quality were determined with respect to lateral and vertical positions relative to Las Vegas Wash and particularly in relation to the BMI tailings ponds and lagoons. Peak concentrations of chloride, nitrate, and TDS are located in areas extending from the plant area to Las Vegas Wash. Figure 9 illustrates the pattern for nitrate which is similar to that for chloride and TDS, although these have maximum concentrations of 3,900 mg/ℓ and 9,400 mg/ℓ, respectively. It is apparent from these patterns that pollutants have migrated extensively from the northern portion of the plant area and from the tailings ponds. This is confirmed by the tritium data.

The effects of industrial waste loading were compared to the mass flux already in the Wash from other sources. At each of three points in the Wash, each point was alternately considered as a "basepoint" where the flux was compared to that at points upstream or downstream. The differential (between the points in the Wash or between the Wash and the tributaries) divided by the mass flux at the basepoint indicates the contribution from ground-water or surface-water sources tributary to the Wash in the reach considered. Using TDS for example, stations LW048 and LW049 (springs discharging industrial effluent) contributed 107,306 and 126,281 pounds/day (48,717 to 57,332 kilograms/day) on March 11 and 30, 1971, respectively. This represents 24 and 31 percent additions of TDS to the Wash relative to the load present upstream from the influence area of the ponds. Considering the mass flux in the Wash above the ponds equal to 100 percent, total contributions

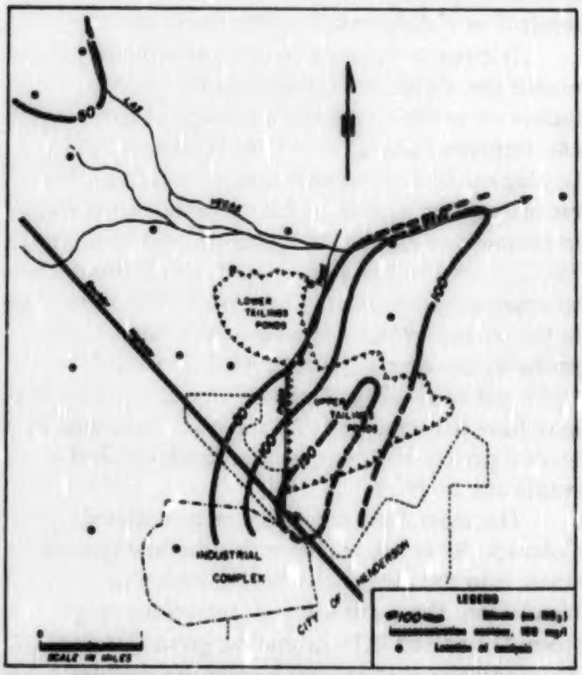

Fig. 9. Nitrate in shallow ground water in the lower Las Vegas Wash area.

from the tailings ponds are as follows:

Parameter	*Increase*
Na	86%
Cl	113%
NO_3	161%
SO_4	65%
TDS	86%

TRITIUM AS AN INDICATOR OF RETURN FLOWS

Of interest to the present study is the use of Colorado River water in Las Vegas Valley in the period 1942 through the summer of 1971. During this period there was rapid growth in population and widespread introduction of tritiated River water for municipal and industrial use. Although the use of such water increased greatly in 1972, the effects of this usage were not recognizable within the time frame of the study.

The effects of tritium concentrations in precipitation over southern Nevada are considered negligible. Long-term precipitation records for Las Vegas Valley indicate that there has been a long-term deficit in rainfall from 1924 to 1929 and from 1942 to 1971. Positive departures from the average occurred only in 1949, 1952, 1955, and 1965. Long-term average precipitation is 4.4 inches/year (111.8 mm/year) whereas pan evaporation is about 90 inches. Except along wash channels and in areas of ponded water, it is unlikely that rainfall resulted in widespread, in-valley recharge.

Tritium in Colorado River water distributed within the Valley since 1961 can be roughly estimated as the straight-line average of the Cisco and Imperial Dam values (T. A. Wyerman, U.S. Geological Survey, written communication). It is estimated that tritium in the Colorado River water imported into Las Vegas Valley peaked at about 750 T.U. in 1963 and 1964, with a uniform decline to approximately 200 T.U. in 1973. Concentrations in the period 1953 to 1963 are little known but probably are on the order of 10 T.U. or less in 1953 and early 1954. Peaks of several hundred T.U. may have occurred in 1958 and 1959 followed by a second period of decline before levels reached a maximum in 1963.

The most dramatic evidence of tritiated Colorado River water recharging shallow ground water is in the area of the BMI complex in Henderson. Here, tritium concentrations range from 212 to 385 T.U. in shallow ground water. This indicates that at least locally the shallow ground-water reservoir contains the probable upper limit for tritium concentrations, considering original concentrations in the Colorado River and decay of 5.5 percent per year. In other words, dilution of return flows has been essentially nil in some areas. In view of recharge from the tailings ponds, estimated to be 28.5 acre-feet/day in December 1971, the extremely high concentrations of tritium indicate little or no dilution of the return flows.

The tritium data indicate waste discharging from the ponds to the Wash is isotopically similar to that from the Colorado River whereas water from deeper zones along the Wash in the reach above the ponds contains 3.9 to 5.4 T.U. Thus, water discharging from depth but upgradient of the ponds, contains essentially background tritium concentration and is unaffected by industrial wastes. The influx of industrial effluents entering the Wash transforms the underflow of the Wash in the downstream direction from older, largely natural water discharging from the Valley, to that which is predominantly young and from the Colorado River.

Tritium concentrations in shallow ground water beneath the urban and suburban portions of Las Vegas Valley average about 50 T.U. and are considered indicative of return flows in the near-surface zone. Concentrations are approximately two times higher in the eastern half of the developed area where Colorado River water, in various degrees of dilution with deep ground water, has been delivered since 1955.

MANAGEMENT IMPLICATIONS

The shallow ground-water system is complex in terms of existing character, past and present impacts of water development, urban land use, and various types of wastewater disposal and return flow impacts. Considering projected water-supply demands (350,000 acre-feet/year, or over two times present water use) and the hydraulic and water-quality responses documented, there is every reason to believe that shallow ground water will become increasingly relevant to basin-wide water management. To date, wastewater management in Las Vegas Valley has resulted in the widespread contamination of shallow aquifers and relatively rapid emergence of return flows either in surface-water courses or as a contaminated veneer at the water table. Past disposal of industrial and sanitary effluent to unlined ponds near Henderson has been particularly unsuccessful. Although at least 230,000 acre-feet of water were disposed of to date, storage calculations show retention of only 3 to 6 percent of the amount infiltrated, which is conservatively estimated at 40 percent. Thus, true waste containment has been minimal and the disposal scheme is an environmental failure insofar as wastes have migrated extensively into adjacent surface-water and ground-water resources over an area of about 16 square miles. Noticeable reduction in flow volume and salt flux is expected to take years, and several decades are necessary before marked change in the total salt flux from ground water occurs after the disposal practice is terminated.

Recharge to the shallow aquifer in the Valley proper has clearly increased with time. Annual accretion in 1943 was 3,800 acre-feet compared to 12,700 acre-feet in 1958, 25,600 acre-feet in 1965, and 38,500 acre-feet in 1973. Recharge to the shallow system now exceeds the total water budget for the Valley prior to settlement. Inefficient lawn watering accounts for much of the recharge to the near-surface zone. If present water and land use patterns prevail, recharge associated with a future population of 750,000 may approach 75,000 acre-feet/year, or roughly three times the total water flux prior to development. More efficient irrigation could reduce the present water demand by 15,000 acre-feet/year and reduce return flows by 11,000 acre-feet/year.

Several policies can be incorporated in management of the shallow ground-water zone:

I. Wastewater disposal (this has been the actual practice in many parts of the Valley for industrial, municipal, and domestic wastewaters).

II. Municipal or domestic water supply (this too, has been actual practice up to the present time in extensive areas of the Valley).

III. Supplemental water supply, for irrigation and industrial purposes (where quality permits and economic development is feasible).

IV. Stabilize, reduce or eliminate saline ground-water effluent to Las Vegas Wash.

The above policies or combinations thereof illustrate possible management directions. Each of these will require tailored monitoring programs as an integral aspect of return flow management. Optimally, the policy and management approach should integrate both water supply and wastewater disposal within the physical framework of the resource, as outlined herein and in relation to the objectives and constraints posed by Colorado River—Lake Mead salinity/eutrophication problems.

ACKNOWLEDGMENTS

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Bull Session on Predicting Physical and Chemical Alteration of Land-Treated Wastewater, and Land Disposal of Sewage

Session Moderator: Jay H. Lehr, Executive Director, NWWA, 500 W. Wilson Bridge Rd., Worthington, Ohio 43085.

Jay H. Lehr, Moderator: Let's introduce the speakers on the panel: Leland L. Mink with Boise State University and on leave of absence with EPA here in Las Vegas at the present time; Donald Runnels, University of Colorado, Boulder; Herman Bouwer, U.S. Water Control Laboratory, Phoenix, Arizona; C. W. Fetter, Jr., University of Wisconsin at Oshkosh; John Mann from Los Angeles; Nils W. Johnson, U.S. Forest Service, Hiawatha National Forest, Escanaba, Michigan.

Rojer Clissold, Hydrogeological Consultants Ltd., Edmonton, Alberta, Canada: Just one question to Nils. I was wondering if you had any indication about boron as a limiting factor in the beautification of the lake?

Nils Johnson: No. We have not found boron to be a limiting problem in our lake. Within the upper peninsula there have been quite a few limnological limitations showing that phosphorus is the limiting nutrient, and we have done nitrogen and phosphorus ratios and have determined that phosphorus is indeed limiting.

W. Roger Hall, W. A. Wahler and Associates, Palo Alto, California: I guess my question is to John Mann. Has there been any research on drainage rates of materials of low-field capacity?

John Mann: There have been some models on unsaturated flow, usually in the wetter climates. I think we have a big communication problem here; at least I do with one gentleman from Maryland who suggested that the tensions could develop in zones below the soil zone to 30 and 40 cm with only gravity operating, and I have a great deal of difficulty seeing how you get beyond field capacity which is generally considered about 10 cm without having some other mechanism coming into play. This is an area where there has not been much work done.

W. Roger Hall: That's been my experience.

Herman Bouwer: Maybe I can comment a little bit on this. This is being researched and worked on really for many years by the soil physicists, and what governs the rate of dominant movement in an unsaturated flow is the hydraulic activity, which is the relationship between hydraulic activity and either tension or water content. You can plot it either way.

Now, when you go to water tension below field capacity, the hydraulic activity is usually several log cycles smaller than its saturation, so for most field soils, say a few cm or a few dm per day below field capacity, you begin to talk about fractions of mm, and in the wetter zone then, in the saturated zone, you can extend unit gradient so that the Darcy velocity, the Darcy downward velocity, is about equal to the unsaturated hydraulic activity. That's a fraction of a mm a day. Your microscopic velocity will be larger because the wetter cross section is less than the total cross section, so maybe that's three or five times larger than your saturated hydraulic activity, or you talk about maybe on the order of a tenth of a mm per day microscopic velocity; so let's say a tenth, that would be a foot per year, it would still be moving at the lower rate.

W. Roger Hall: Relative or percentagewise, does movement in the vapor state become important in relation to movement in the liquid state?

Herman Bouwer: Eventually you will get only vapor movement, but when you got to that state, I don't think you have to worry about ground-water condition; but as long as you are above the hydroscopic water, you will have water movement in a liquid phase and you will have downward movement. Now, in a few years you may not detect much. In a few hundred years you might be in for a surprise.

John Mann: This to me is a very important thing, whether you say that the water is able to move down and then somehow completely stop. If you adopt the concept that the movement only slows to a very small rate, but that, in fact, movement goes on regardless of the rate for centuries, then you nevertheless are faced with the problem of some type of water pollution or some potential for it.

Now, this may be theoretically true within the time frame of lab experiments, but as the water moves down and as the amount of water available at the lower levels gets less and less, I still think that there is such a thing as a pellicular front when you put water on the dry desert soils, and these are dry. I have been involved in the bucket-augering of many places in the creosote bush areas, for example, and this stuff is absolutely dry. I think the important thing is that perhaps the movement Mr. Bouwer suggests does go on, but that other processes begin to take over. Now, for example, the work done at Hanford by Isaacson and his group indicates that vapor transfer is not an important mechanism. The temperature control from season to season has more influence on the moving of water up and down, primarily. Perhaps it is slow, but perhaps mostly in the vapor phase, so I think that regardless of how you approach it, the concept that the water continues to move down through these dry vadose zones even at an extremely small rate, is important.

Now, if you start thinking in terms of centuries you don't have to have a very high rate to conclude that it gets there, but I think what we are faced with here is the interposition of a number of other mechanisms which will not only control, but perhaps even reverse this very tiny tendency for water to continue flow down in the partially saturated zone.

Donald Runnels: Can I say something on that? Not being a hydrologist I keep looking to hydrologists for wisdom on this, and I am constantly amazed at how little is known about it. Bill Guyton proposes piston flow. He proposes that the rain that falls on the surface mechanically shoves down the water that—pellicular water, the water on the surface of the minerals well below field capacity. No gravity flow. You have to wait until the next rainstorm comes along and put a gradient down on this thing, and rather than the new water draining through the interstices between the grains, it pushes down the old water that was there before in a mechanical piston-flow fashion. Guyton is a man who has worked in arid regions for a long time, and he is still guessing. I am amazed at how little is known about it.

Herman Bouwer: I think the piston flow holds true only where there is fairly high water contents.

Donald Runnels: An important factor would be, I suppose, compressibility. The water is, you know, not easily compressed, and I suggested that this thing is mechanically transmitted down sort of as a wave.

Herman Bouwer: I think in the top material, you have a process of infiltration and redistribution and diffusion. Eventually it might come down to the deeper depth.

John Mann: There is some evidence of this following very heavy rains, and with an increase in barometric pressure there have been some evidences of the increased barometric pressure pushing on the saturated water in the zone, compressing the air underneath it and transmitting a small amount of that differential to the water, only in the water-table situation directly, and we are dealing with relatively wet areas and relatively short, partially saturated zones. Now, it's almost as if you were to have a flexible membrane on top of the system that the air pressure above increasing, pushing on the air underneath, and that being transmitted to the water table. I think we are talking about very small magnitudes of movement, and also rather specialized types of conditions.

Donald Runnels: Do I draw the conclusion that we don't know much more about water in the vadose zone?

Herman Bouwer: I think if you ask the physicist, he will agree that they know quite a bit on the theory, but when it comes to applying the theory to an actual situation it's the difficulty of getting a proper input data that prevents him from making intelligent judgments. Its relationship to the saturated hydraulic activity or water content or tension can be determined in the lab on samples, but to do this on field soil, natural conditions and then for a large profile, say the entire vadose zone, it would be very difficult. I think the general principles are known.

W. Roger Hall: In applying this practically for waste disposal, with the current state of technology, what kind of a safety factor do you think you should apply for, say, disposing of potential contaminants? In my case I am not suggesting very hazardous wastes, but, say, high TDS water, for example.

John Mann: I think on a high TDS water you can tolerate a much lower safety factor than you could where there is some particular toxic element. If you are dealing with the normal, merely elevated TDS saline solution with normal ions, it will depend somewhat on the rate at which this material is redissolved, and assuming that many of the reactions that Don Runnels has talked about do not permanently tie up these things in a way that they are not easily redissolved with just one wetting field capacity, you may not be able to get the pore volume flow through. And so I think the net effect of this is, if we returned to an ice age and you had a reinstitution of deep percolation of rainfall, the amount of resolution of this material would be relatively slow.

Donald Runnels: I am more concerned, Roger, with highly toxic contaminants, and our philosophy has been that unless they are tied up, regardless of a very large amount of water passing through, we have to worry about them, and perhaps they shouldn't be discharged at all. Take arsenic, for example. Let's test the soil. Let's test the alluvium or its capacity to remove arsenic irreversibly in essence against many events of precipitation, and then we can be safe.

TDS is a tough one because if any water gets through, the TDS will probably get through, too; so I'm on the other side of the spectrum. There are highly toxic contaminants and they have to be removed and tied up.

Jay H. Lehr: I have heard two different things at the Symposium with regards to desert hydrology. I clearly had the feeling at our first session that while we don't know a great deal about all the things that happen as the water moves down from the surface to the ground water, we do know that the desert seems to be a better place to dispose of liquid waste than in a humid climate, because it is slow and there are good things that are likely to happen, and that we can use the soil to treat waste.

Then last night at the Las Vegas Valley meeting I didn't get the idea that they felt the picture was very good at all; that is to say, taking the theoretical approach that we have taken at the first session, and the practical problems they have here at Las Vegas Valley with the growing population and poor planning with regard to their disposal of waste, they didn't seem particularly buoyed up by the fact that they are here in the desert rather than back east.

John Mann: Jay, I have done some work here in this area, and this is one of the world's most horrible examples, and I don't think we want to be too influenced by this particular situation, because it is perhaps one of the worst ones, not in terms of the toxicity of the material, but in terms of just the huge volume of wastes, the high salinity of the waste rather than the true toxicity. Industries were developed in World War II which had very large flows of water. This was done under wartime conditions where no consideration was given to disposal of wastes. It was gung-ho for the war effort in a situation geologically that was not too well known or understood, and so we have now the heritage of these conditions developed during a wartime push. The slowness of movement of these materials once underground is just continued for decades. I think if there is one thing in common in this type of pollution it is that you can get it in the ground relatively quickly, and it takes a long time to get it back out.

In addition to that we have superimposed upon these rather thoughtless industrial disposals, an enormous growth in domestic sewage, particularly municipal sewage, and in a hydrologic or geohydrologic environment where everything is funneled down one little wash. We have 40 million gallons a day, high in nitrate, high in phosphate, dumped into a river which shows a rather sharp increase in salinity. We have here one of the horrible examples that we can use as a gauge against what we should do next time.

Ronald L. Barto, James M. Montgomery Consulting Engineers, Inc., Pasadena, California: In talking about this vadose zone, it seems to me that evaporation is going to take a lot of, shall we call it, short-circuiting of the water on its trip downward, and evaporation will cause a lot of this water to move back to the surface and never make it to the water table. I would like John to comment on this, particularly in arid regions where we know that the field capacity is so low. Does this evaporation go all the way to the water table, particularly if the water table is, say, a hundred or two hundred feet deep?

John Mann: I think there must be some sort of vapor transfer. We know just from the fluctuations of water levels and with barometric changes that there must be quite a bit of air flow through the soil, and certainly with the flow of heat in and out this will also have an effect. I personally feel that the evaporation from pellicular films in desert regions is certainly one reason why moisture levels are extremely low. Somewhere in the past there was a colluvial period. There was a time of old lakes. Many of our valleys, Death Valley, the hottest place in the U.S., had a few hundred feet of water in it. We must have had some rain passing directly through the soil all the way to the water table. Since the end of this time, there has been a progressive removal of moisture between those materials and the water table. The mechanism by which this occurred is very difficult. I think airflow evaporation is one of them. Probably heat flow is another. Then when we speak in terms of thousands of years we get to the range in which soil zones would be developed. We have a lot of feldspar minerals in our desert soils, and the feldspar minerals particularly will hydrate and tie up a lot of water as they become clays. I have seen materials where you can still see the shadow of the cobbles. These cobbles, which probably were granitic to start with are now turned completely to clay. They are imbedded in a clay matrix, and they are all clay, and you can see this from the top of the zone down to fairly fresh material in the lower part, and that has to be associated with the tying up of water. The main source of water in the weathering below the surface in a dry environment, would be the pellicular films.

Donald Runnels: I would like to make just one very brief comment about the downward movement, and then loss by evaporation. I think the presence of zones of caliche in desert regions today are showing us about how deeply most of the water penetrates before it is lost. Mass balances that are done on the caliches say that most of that calcium has to be coming from the surface, from windblown dust, and is moving down, and if you find a zone of caliche that is four feet below the surface, that suggests that the water is stopping at about four feet below the surface before it is lost again by evaporation; and so I take it as a good sign for a place to dispose of wastes, a place where we have good thick caliches forming today.

Ronald L. Barto: Then you are suggesting that maybe 25, 50, or 100 feet is much too deep for evaporation to take place, is that right?

Donald Runnels: No. I am thinking of the recharge of the occasional precipitation event then being evaporated back up, but I can't imagine why there shouldn't be evaporation in a water table in desert regions. If we have a chemical gradient from pure water at the water table to 10 percent relative humidity at the surface, it seems to me that over a long period of time you simply have to have evaporation. I was addressing the occasional rainstorm.

C. W. Fetter: It seems to me there must be regions where, because of natural processes, the soils are unsuitable for agricultural purposes and the ground water has such mineralization that you can't do anything more to harm it, it's beyond use. Is there a potential for looking for areas like this and then saying, well, this is a good area for underground waste disposal because even if we do have movement of these dissolved substances, even if they are toxic, the ground water is toxic, the soil is saline, we might as well use it for a disposal area?

John Mann: Yes. I have gotten myself at cross-purposes with certain regulatory agencies in suggesting that waste disposal is a very important part of our existence, and I have proposed even in some instances where an alternative water supply is available, that certain now-fresh bodies of ground water be dedicated to waste disposal. This might be an area in a desert basin, usually where the water cannot flow readily to other areas. We do have in most of the closed basins, the Mojave Desert, the Antelope Valley area very good examples of this. Within this system there is a good deal of recharge from the San Gabriel Mountains. There is flow through a series of basins down to Koehn Dry Lake, which is the sump for the whole system. The water there now is probably 10,000 to 20,000 ppm. This is the sump of the natural system. All of the salts from this system eventually will wind up there because there is an integrated ground-water flow down to the bottom of this. There is an isolated fault basin. All the salts build up there.

The costs of waste disposal are becoming so high that we can't divorce the problem of, say, trying to save maybe $1 million worth of fresh water when you can get a $10 million benefit out of using this basin for waste disposal, assuming, of course, that you have another source of water.

Waste disposal should be viewed as one of the important economic elements. Designation of Class I waste disposal sites in California means we can put highly toxic elements there. Finding a spot like this that will supply all of the rigid criteria is very difficult. I drove way back in a mountain range on a dirt road and found a little dry lake in a closed ground-water basin with internal drainage. I wanted to designate it as a Class I site, but it turns out that there is a Bureau of Land Management grazing lease on the area. To my knowledge there hadn't been any grazing or grass there for 30 years, but in the early forties there were a couple of wet years and grass grew. Now, there is an example of a wasted resource. Class I waste disposal are disposing of toxic substances, things like Kepone, where you can set it out of the way so it isn't going to create a problem to existing facilities.

Jay H. Lehr: This Kepone situation strikes me as being an example of what can be done, and it can be done easier out here than in humid Virginia. That was almost a military operation the way they dismantled and buried it. Out here in the desert we have areas where we could be doing that. Why don't we?

John Mann: Well, as the great champion of the desert environment, I don't want Kepone here! I don't want any problems like that here—let's use Ohio! (Laughter)

Charles P. Vanderlyn, U.S. Environmental Protection Agency, Washington, D.C.: They mentioned on the field trip on Sunday that they were thinking about using the test site area for radioactive waste material. Do you have any comments on that?

John Mann: Yes. I will refer you to Ike Winograd's paper. He has suggested putting the waste in containers and putting them in the bottom of holes within the vadose zone. He has gravel sheaths around these so that hydraulically any water that might come down gets diverted off to the side away from the container. This is certainly worth looking into, because one of the biggest waste problems of our age certainly is the disposal of nuclear wastes because of the long half-lives. I worry very little about the ordinary ions that we find in drinking water. It is somewhat like Bill Walker pointed out this morning, which I am sure brought gasps of horror to certain people, when he said they should have taken the cyanide and let it run into the creek on down into the Mississippi River. I gasped a little, too, at that, as a matter of fact; but I could see if the thing were done intelligently and were released with a knowledge of what the dilution factor was, it could be done safely.

We have three types of wastes: ones where the only thing is sort of a high salinity of the normal ions, and this poses perhaps over a period of years the least type of problem; the second is where you have toxic substances, you have to be more careful; and the third and most difficult are the radioactive wastes because of the half-life.

Donald Runnells: In our proposed regulations in New Mexico regarding the previous question, we proposed not to protect water that has TDS gradient of 10,000. Perhaps the best use for that is disposal. In talking to environmental people I have brought up this possibility to them about disposal in the vadose zone, and it really follows Winograd's proposal about getting rid of radioactive wastes. I presented the picture that perhaps he can dispose of metallic elements today that are problem materials, lead, copper, arsenic, and wouldn't they like it if 10,000 years in the future that material was still contained in one small area in solid form and could then be excavated, if necessary, and reextracted. Ten thousand years from now that might be a resource, and somehow this strikes a favorable chord with my environmental friends; if you have to get it out of the vadose zone, you can.

Jay H. Lehr: Waste has been described as just substance without a use, you know. One can say there is no waste. It is just matter that we haven't found a use for yet.

Donald Runnels: In the vadose zone in an arid region, we wouldn't even have to encapsulate some of these wastes because nature would do it for us chemically.

Jay H. Lehr: Could not a great industry develop in the desert, in the area of solid waste disposal? I mean, could we not take thousands of acres of useless land and begin disposing of the nation's solid waste in some type of vault or nonvault?

Richard H. Pearl, Colorado Geological Survey, Denver: This is a very cost-intensive procedure, and companies can't afford that. For example, we have one there at the Denver Metro area where Shell Oil Company has been making for years this pesticide at the Rocky Mountain Arsenal. They were ordered to clean up the water problem they had there. They wanted to put it out on the Lowry Bombing Range right over one of the major aquifers of the Denver Basin. A lot of people raised objections to that. Well, there was a disposal industry in Houston, Texas, and they were willing to take this stuff, but Shell Oil couldn't afford to ship it to Houston. They proposed to put it in deep-well disposal, 5,000 feet deep. That was turned down by the State. Today it's still being buried out there over that aquifer.

Herman Bouwer: It is the low rainfall and the aridity of the air that makes the desert attractive for waste disposal. In some of the papers that were presented these last few days on the garbage dumps and the landfills, the main reason you have the leachate production in the eastern States is because of the high rainfall. In the desert, sanitary landfills are no problem because there is no rain, there is no infiltration, and no leachate production. Maybe we should have indoor garbage disposal areas in the East. Keep the rainfall out, don't produce any leachate. Put a roof over your sanitary landfills. You can collect the rain water on the roof and use that for potable purposes.

Richard H. Pearl: You can do that. A lot of sanitary landfills are being constructed today with impermeable barriers on all six sides of them, and even on top. Hopefully it is going to work and keep the moisture out of it. Any rainfall that falls on the landfill in the filling stage is collected and moved out of the area.

Ronald L. Barto: To illustrate the fact that you need to keep the disposal sites reasonably close to where the material is being manufactured, we are working for a client in California where a Class I disposal site was closed due basically to political reasons, and they were unable to find another site to reopen, and as yet they are still looking. For the past three to four years most people feel that probably where these Class I wastes are being deposited are either along the roads in Southern California or in the sewer lines, because none of the other waste disposal sites in adjoining areas have been receiving this waste. So if you make it too costly to transport this waste to adjacent areas, the producers will find other means to get rid of it.

Nils Johnson: In designated desert areas and disposal areas you would require complete reconsideration of a number of industries. Industries, I would think, would tend to move to the area where they could most easily dispose of their waste and yet be near a usable natural resource to use in whatever products they manufacture.

Jon O. Nowlin, USGS-WRD, Carson City, Nevada: I would like to make a couple of different points. One is the transportation issue. Right now, one of the reasons that steel is not recycled more often is that there is a transportation rebate in transferring iron ore from the mine to the mill. There is an incentive in terms of the tax structure that doesn't exist for scrap, so if you can't take scrap cars and within the same State, recycle them back to the mill, you are not going to be able to take wastes from Nevada economically and ship them back out east.

No. 2, as a recently immigrated Nevadan, I would like to defend our State against the people who want to use it as a dump for the nation, and show why people are against it within the State, and that is because of examples such as Las Vegas.

No. 3, I think that we will take a giant step towards coming up with the proper solutions, both politically and technically, if we talk more of waste storage instead of waste disposal, and get to this idea of taking some of these materials that are now hazardous and let's talk right from the start about storing, not disposing of them.

Charles P. Vanderlyn: There is tremendous energy potential in our solid waste. I know in Germany, 20 percent of some of the cities' electrical and generating capacities is in recycling of the cities' ordinary wastes.

Russell E. Darr, Wright Water Engineers, Denver, Colorado: I can give you an example of just how people are afraid of something new. There is a certain brewery in Colorado that was looking for various municipalities in the Denver area to take their garbage for nothing to generate such a plan as you have proposed in energy production; and they finally gave up when they could not find a municipality which would donate their garbage, so you have got a political problem to overcome before you can look at possible scientific and practical aspects.

Leland L. Mink: We have just completed quite an exhaustive study of the Boise-Caldwell-Nampa urban area in southwestern Idaho on this idea of taking collection of garbage, and one of several alternatives, there was a proposed coal-fired plant. The industry was interested in utilizing all the combustion material in this, but we could not talk local governing agencies into even considering collection and transportation to that State. Political reasons again.

Charles P. Vanderlyn: How do we as technical people surmount the political problems involved in coming up to solutions?

Jay H. Lehr: I think the answer to that question is the development of continuing communications between the scientific communities and what they are doing, and the public. And now our time is up.

Bull Session on Controlling Pollution from Sanitary Landfills, and Reduction of Nitrate Contamination

Session Moderator: David Miller, Geraghty & Miller, Inc., 44 Sintsink Drive, Port Washington, N.Y. 11050.

David Miller, Moderator: Our panel speakers are: Joseph H. Baier, Suffolk County Dept. of Environmental Control, Hauppauge, New York; Lawrence A. Eccles, U.S. Geological Survey, Laguna Niguel, California; Todd Giddings, Todd Giddings and Associates, State College, Pennsylvania; James L. Mang, SCS Engineers, Long Beach, California; Michael A. Apgar, State of Delaware Div. of Environmental Control, Dover; and W. B. Wilkinson, Water Research Centre, Buckinghamshire, England.

Ken Childs, Suwannee River Water Management District: Mr. Baier, the two years of data suggested that the nitrates were highest beneath the sites that had lowest application. And then you concluded by saying that, however, based on six months of data that wasn't presented today, we feel that the original conclusion was true. Can you explain why the first two years of data showed the other trend and support this?

Joseph H. Baier: It wasn't two years of data. It's only what I showed in the overheads. There was only one year of the data, from March 1975 through March 1976, which took you through one full growing season and just up to the start of the second growing season. The only background data we had on the nitrogen was our first samples as soon as we put the wells in, which was approximately the first piece of data. We have no idea what the historical water quality results were on each of the demonstration plots. Why one was higher than the other I don't know. Now, the second planting, which took place this year, 1976, and is just being harvested now, we continued with the monitoring with the same number of wells and we put in more wells at each site. We only had one well, which is approximately centrally located on each of the plots, with a grower practice and the recommended practice. We then added another well upgrade or just immediately adjacent, off-site, to tell us what the background was coming on to the site and we added another well on each half in the plots themselves to verify that we were getting the same nitrate data in two spots. This year's data is now showing that the reduced nitrogen application loadings is yielding better ground-water quality in the recommended plots over the grower plots. In other words, those graphs are now reversed. The trend started high and then started back down on just about all of them. They went up and stayed up there for most of the growing season, but they were coming down, and that downward trend for the recommended practice in 1975 was statistically significant and it continued on into 1976.

Ken Childs: So you think this relationship is one of time delay from application to results?

Joseph H. Baier: Yes, you've got certain mineralization. You've got organic matter that's built up in the soil for years in the root zone and that has to adjust. The amount of nitrate, which is being mineralized and leaching out—there has to be adjustment between the soil interface, the crop interface, the fertilizer application and ground water, and I think we saw that. While the crops responded pretty much as we had hoped, ground water took a little bit longer.

Charles Kreitler, Bureau of Economic Geology, University of Texas at Austin: What percentage of that nitrogen that you feel is going into the ground water is the result of mineralization of the organic nitrogen in the soils versus direct leaching of fertilizer into it?

Joseph H. Baier: Part of the study included a detailed review or nitrogen budget of fertilization as to what could happen to all different forms, the plants themselves, the plant itself which is plowed under, the organic material that is still remaining in the soil, the possibility of volatilization, the possibility of runoff, losing some of it because of that running off the site, and many other features throughout the entire nitrogen budget for a potato farm. And the fellow who did it was Jack Meissenger from Cornell University. He came down to three basic materials as controlling the whole situation. And as the nitrogen loss is equal to the nitrogen applied, minus the nitrogen in the tuber which is harvested, very simple balance for the whole system, everything else came out as being negligible, too small a quantity to really quantify or to really have an effect upon the losses that you were talking about, because we're talking close to a hundred pounds of nitrogen per acre lost. But he did go into details on mineralizations, the amount of organic material present. He discussed that some of the growers also plant the rye crop, cover crop for erosion purposes, just to keep the topsoil there in the winter. And they also add fertilizer to that. It's an additional loading at the end of the year to get the grass to grow quickly, and this all adds into the input.

Charles Kreitler: I've done some of the nitrogen isotope work on Long Island. Steve Ragone and I are sampling. We see some interesting problem in that the nitrogen isotope ratios for the ground water in Suffolk County are running in a range of approximately plus three parts per million as plus seven parts per million, and the fertilizer in general is down around zero parts per million. Somewhere we're getting an isotopic shift that is causing our ground waters to get isotopically heavier than the nitrate fertilizers. Now, we see this in several other aquifers, and the amount of shift appears to be dependent upon the type of fertilizer we are using. If you use a urea anhydrous ammonia, you get a further shift. Somewhere we're changing the isotopes in this system. Somehow I'm wary of the concept that the nitrate going into the ground water is equivalent to the nitrate which was put on minus the nitrate that went off in the tuber.

Joseph H. Baier: You're saying, in effect, there would be a direct interconnection, that amount that's applied goes right through?

Charles Kreitler: If there is a direct flushing of the nitrate from application directly into the ground water, we should expect to see that take isotopic composition. Ground water is going to be identical to the fertilizer and we don't see this. And I don't know where the problem is.

David Miller: Maybe for some of the people who don't know some of the work that you've been doing, you could explain it—some of the work you did in Queens and Nassau.

Charles Kreitler: Right. The paper in *Ground Water* two years ago, I guess it is, we looked at natural nitrogen isotopic variation of N 14/15, and we are able to differentiate different isotopic ratios for nitrate sources that originated from animal sources versus cultivation phenomenon. We applied this technique to some areas in western Texas, where we had very bad contamination. We could identify the major source of contamination of organic nitrogen in the soils due to cultivation. Enough fertilizer used, but we could separate out the animal waste. We did not see the isotopic signature there. Since then we have looked at a number of different aquifers and in areas where we have minimal cultivation, but, say, heavy dairy farming. We see that the nitrate in those ground waters is related to the nitrates coming off those barnyards. We just finished a study on nitrates on Long Island, around 20 samples. You can see that when you look at Suffolk County you have one isotopic range, and that's Pleistocene there, and you move towards New York City and you have the deep Magothy aquifer. That has a slightly heavier isotopic range. The Nassau Pleistocene is an even heavier isotopic range, probably indicative of septic tank contamination there. And Queens County supposedly never had much in the way of cultivation at all and was based on septic tanks and leaking sewer lines, and it has a much heavier isotopic value.

Unknown: Mr. Eccles, if you determine any interconnections as they do exist in Los Angeles between the upper aquifers and lower aquifers and you came up with some face rate, pumping from the lower aquifers so they would not be contaminated from the affected aquifers on the top either through the interconnections themselves or through any abandoned wells in the area which you described yesterday.

Lawrence Eccles: At this point we do assume that the upper portion of the aquifer is connected to the lower portion, but you don't necessarily get recharged to the different portions of the aquifer from identical spots. The lower part of the aquifer may be recharged from an entirely different source than the upper part of the aquifer. The well number two right next to well number one obviously is an interconnection. Some water's going to flow downward in well two through the ingradient and just by well one flow down, of course, into well one. We'll probably recommend that they plug well two up.

Unknown: My immediate concern is that there will be greater pumping from the lower one, if that would affect the gradient and localized recharge in the area itself from areas which are contaminated with the nitrates.

Lawrence Eccles: This isn't something we've thought about just yet. That would be part of the greater basin management plan, to recommend what the yield from various zones in the aquifers would be, and one thing that might work would be a pump test.

W. B. Wilkinson: Mr. Baier, what was the thickness of the unsaturated zone and have you been able to measure any nitrate profiles down through the unsaturated zone or any tritium profiles? I'm a little concerned about the interpretations you've made on the basis of the nitrate concentrations in the saturated zone in the ground water. It could be that the rate of movement down through the unsaturated zone is very slow and that what in fact you're observing is a land use or a fertilizer application that may have occurred possibly several years ago.

Joseph H. Baier: Specifically with respect to the demonstration plots that we selected, these plots were regular farmed areas and have been farmed areas for 20-some odd years. The only change would be occasionally the farmer would not plow the field and just leave it for a year just with no crops whatsoever or once in a while he would rotate a crop, just to change the over-all composition. The depth of the unsaturated zone varied from 2-16 feet. We did this specifically because we wanted to be able to pump the wells themselves from suction, using suction pumps. The wells were installed with skimming wells right at the top. The type of infiltration on Long Island is coarse sand and gravel. You're talking about infiltration rates in the neighborhood of feet per day. We have very little or hardly any runoff, regardless of size of storms. If you would travel through Long Island in a two or three-inch rainfall, you'd see puddles all over the place. Two hours after the rainfall had ended, the streets would be dry and no puddles on the side. Everything has seeped in, so we know that it goes right through. We explored tritium, but the fact that we were actually using farmers' fields and he was going to have to harvest the crop, regardless of the fact that tritium is harmless, I don't think we could have talked him into using it.

W. B. Wilkinson: I wasn't talking about the application of tritium. I was talking about the measuring of tritium that's in the rainfall as a result of thermonuclear explosion from 1953 onwards.

Joseph H. Baier: We had explored the possibility of applying some just to see if this infiltration was directly looking at the background, then applying a known quantity and seeing what the result and reaction would be on the ground water, since we knew we had fairly direct connection between the land surface and the ground water. But there were a lot of logistical problems there.

Ken Childs: Now, I have listened to you talk, Mr. Giddings, and what your conclusion was essentially, was that site suitability may be somewhat lacking but the installation of a "plastic liner" guarantees the integrity of the site. This is really the bottom line in an awful lot of proposals for landfills today. In addition, I'd like to say that there is a recent E.P.A. document suggesting the questionability of liners. Now, can you demonstrate with some existing study or documentation what the lifespan of a liner is, or a different kind of liner and what the failures are?

Todd Giddings: The suitability of geology at the site and the example I gave is an important aspect of the liner installation and performance at that site. I think you can see where the four natural backup features in the site made the danger of leachate contamination of ground water minimal, if not virtually impossible in the event of a major liner failure at this site. So I was pointing out the utilization of a marginal site through the installation of a liner and further designing the installation of that liner in the operation of the site to take advantage of the natural site characteristics. I think it has to be in harmony. I agree with you, if you go into a very bad geologic site and say because I've lined it I'm okay, you may find it impossible physically or economically to correct a leak should it occur. The P.V.C. liner was chosen only after an exhaustive review of literature and research on many types of liners, such as hypalon, E.P.D.M., monimer, butyl rubber, petroleum drive, petromat types and asphalt, and was selected based on the available state-of-the-art in researching liner performance under attack from "leachate." And the reason I put that last word "leachate," in quotes, is because if you look at published chemical data on the constituents in leachate, sometimes the ranges are two or three orders of magnitude for concentration of critical parameters. So there's leachate and there's leachate. As far as a leachate attack, there have been studies and there's literature documenting satisfactory performance under attack from strong leachate. Since we're in a new technology, we don't have 25 years of data. There hasn't been P.V.C. under a landfill for 25 years anywhere, so, unfortunately, the tests sometimes are accelerated under extreme lab conditions. Other field tests have been as long as 2 to 5 years, and so we are in an area of chemical extrapolation, but it has been addressed in the literature and we feel that since that's a very important question, the soil layer immediately below the refuse will become saturated with water and provide a capillary barrier to contaminants that would move down out of the fill that aren't admissible in water and hopefully would perch above the liner. And the sand layer above the liner has a permeability based on the leachate we expect because we want to maintain an unsaturated situation above a liner. We don't want leachate sitting on a liner to have the time to attack it; thus, with regards to the point you raised (what about the durability of a liner) we put those features into this particular multilayer system. Because I agree, those points you've raised are important. They are not fully known, and we had to approach them in that way.

Joseph H. Baier: How do your various respective communities feel about resource recovery, as, let's say, an alternative to landfills?

Ken Childs: I don't think you're going to have any success with the recovery of garbage, or whatever, until there's some way to support it economically, and right now I don't think you can recover these materials economically. In north Florida there are programs of aluminum can collection centers, but that's, you know, just a drop in the bucket. I don't really see it until it becomes economical.

Joseph H. Baier: You can recover paper. You can recover the glass. You can pick up other metals besides aluminum. I'm talking about almost complete resource recovery so that you're left with basically garbage, and then you incinerate that.

Randy Sweet, Consultant, Kelso, Washington: I know of a county in Washington where they were funded by EPA for a grinder. It's operating today. The price of disposal of garbage, which they now bury, went from $2.50 a ton to $8 a ton and they're still burying the stuff. The marketability of cardboard became a very big market in the northwest and immediately everybody wanted into it, and a lot of stores put in their own baling equipment. The market went well over $50 a ton. For a long period of time everybody was making money and then the bottom fell out of the market and the bailers sat around. The main market that people think they can depend upon is the fuel market, and right now private industry in Oregon is putting together a grinder. They've developed a grinder for tires and I think that's the way it will go. They'll pick out particular fractions which are easy to separate and hard to dispose of and high B.T.U. values and use as a fuel.

Jim Mang: I have a comment on your resource recovery considerations. Most of the statements have been directed at direct resource recovery. Sanitary landfills can play an important role in indirect resource recovery due to the installation of wells and the extraction of methane. This is a feasible alternative, depending on the size of the landfill and another, a variety of other considerations. In small landfills enough methane can be recovered, say, that a golf course is built on top of it to run the air conditioning, etc.

Ken Childs: Mr. Mang, to quote your paper, "expensive and not entirely effective are plastic liners," and then you make a pitch for clay liners. Is there documentation to back up "the efficiency of clay liners," and can you address clay liners simply on the basis of personal ability or do you have to address mineralogy? Don't you feel that it's necessary to address the length of time that it will take the leachate to move through that liner and the chemical reactions that will take place between the mineralogy of the sediment, including the clays and the leachate itself?

Jim Mang: I didn't make a pitch for clay liners. I think the state-of-the-art is an area of flux where there's not that much specifically known about either clay liners or P.V.C. liners and their longevity. I think it's been demonstrated that P.V.C. liners won't do the job and the potential exists for clay liners, and I think this is the only extent that anybody can speak of in terms of longevity. Clay liners in my opinion offer the most promise because they are a natural soil product. What you have to do is utilize the clay liner to force the leachate to float at a given area, then get it out of the landfill. Leachate or certain contaminants in leachates, some of the heavier metals, will migrate through the clay and it is a function of what is required by the regulatory agencies on what you can and can't do.

Todd Giddings: I have to respond to your statements where you say that it's been demonstrated that P.V.C. won't do the job. I can cite you instances where I could demonstrate that steel bridges have failed and therefore I can say that steel won't do the job and we should stop building bridges out of steel. Well, that's patently absurd, as is, I maintain, the generalization that because P.V.C. has failed in certain applications of landfill liners we should cease lining landfills with P.V.C.

Jim Mang: I think your permeability differentiation is your most ardent safeguard, but I just cannot see spending the money on a blanket basis for P.V.C. liners when nobody can demonstrate that they'll last a long period of time. I wish you could show me a situation where they have lasted over more than 20 years. I'd be glad to look at it.

Todd Giddings: Wilbur and Orville Wright wished there had been someone else do it first, I'm sure, too. I'm not advocating the general widespread use of P.V.C. liners as a panacea for landfill site design. In fact, I'm a proponent of natural flow system manipulation and control as part of collecting leachate in a landfill site. And we in fact have sites in Pennsylvania and elsewhere that are operating without any type of liner, using the natural flow system, manipulating that flow system and collecting the leachate.

Jim Mang: In your talk I could not understand why you had the P.V.C. liner until you explained to me that it was a cost-effective situation, reducing the flow of ground water and reducing the flow of leachate. In that sense I could understand it, and only in that sense.

Todd Giddings: The site again is in a ground-water discharge zone. To place the refuse in contact with the ground and to allow that water to seep into the base of the fill would greatly increase the quantity of leachate generated, and in fact you can visualize them thinking back to that sandwich situation, draining the ground water from under the liner keeps it out of contact with the refuse. And by reducing the quantity of leachate generated, we realize a great savings.

Randy Sweet: I was wondering how you put together your plumbing system. I believe you said 140 feet of garbage stacked up in there?

Todd Giddings: 120.

Randy Sweet: Okay. A large face. One of the biggest problems that I've run into in landfills is what I refer to as the "nuisance seep" all along the face, one or two gallons a minute in a multitude of places. How are you separating that dirty water from your clean water so you don't end up treating all the runoff from the landfill?

Todd Giddings: The gas vents which will be constructed throughout the landfill—several dozen of them—will consist of precast sections of perforated concrete pipe, culvert pipe on end, packed with gravel with a plastic or perhaps steel pipe within the center of the major gas escape avenue. These pipes will be added to and built up as the fill depth increases. They're going to be the short circuit drain. They'll be connected directly to the leachate overdrain network and they'll be the short circuit to bleed off the perched water, because we've recognized exactly what you've pointed out, that the nuisance seep of leachate breaking out of the front face of the landfill due to perching on our intermediate or daily cover is a problem.

Randy Sweet: Have you looked at orientation underlifts at all as far as moisture routing system within the landfill? Not an entire lift, but the daily lift in the compaction.

Todd Giddings: We haven't specifically looked at the orientation of the working face, but I recognize what you're saying and would thank you for a very good suggestion with regard to creating an anisotropic to be favorable to prevent the breakout that we're trying to prevent.

Unknown: How are you going to treat the leachate?

Todd Giddings: Initially, the leachate which comes out of a drain system will go into a holding lagoon of approximately 1,000,000-gallon capacity, in which there will be floating fan aerators. After aeration, that treating leachate will be piped back on the landfill and recirculated to accelerate decomposition of the refuse and stabilization of the mass. This is along the lines of work done by Poland and Hammond and others that's been reported where recirculation accelerates the leaching process, because we're aware we face the problem of how long are we going to treat the leachate. We've done water balance calculations and there will be a point where we'll have reached field capacity and have more water coming out than we can recirculate back to the fill. At that point, we'll have to look at what the quality of that is with regard to what type of treatment we'll have to construct at the site and then we'll have a discharge to surface water under a State permit. Money will be set aside in an insurance fund for just that type of endeavor, to keep that cost down. We're going to recirculate initially.

Unknown: This process continues on for a hundred years. It will be exhausted and there will be a problem for the children.

Todd Giddings: I beg to differ. Based on research work by Poland and Hammond, they show that the major constituents in leachate in recirculation situation begin to decrease to acceptably low levels for a low level of treatment or no treatment.

Lawrence Eccles: If the integrity of the liner is in question, why not just use an additional liner, use two or three if you have to? They're relatively inexpensive, aren't they?

Todd Giddings: And by using the probability of reasoning, if you double it, you multiply the probability together, thereby achieving a greater safety than just a factor or two?

Lawrence Eccles: No, as time goes on, the leachate is going to become less potent, so maybe it will react with the first liner over so many years. Well, have a layer of clay beneath the first liner and then a second liner. At least by that time we'll be digging the garbage up to reprocess it anyhow.

Todd Giddings: In our system, we have the backup naturally occurring features beneath the liner, the underdrain, the ground-water discharge, the permeability contrast, the artesian head, even beyond that, the bedrock with the monitoring wells giving us an extraction capability that we felt it was not cost-effective to go to two liners. We had the natural features at our site in lieu of a second liner, but I could visualize a site perhaps on Long Island or a major aquifer where that protection might be warranted because I agree it's unknown as to when that liner will finally fail.

Ken Childs: There's something I don't understand. If effectively you're developing a closed system in your landfill and you recirculate the leachate, how do you end up in 20 years or 40 years with no leachate?

Todd Giddings: We definitely end up with leachate. We performed a water budget or water balance, a series of calculations, taking into account precipitation, infiltration, runoff and the moisture deficiency of the incoming refuse, and generally I think it's been found that a foot of refuse has in the neighborhood of an absorbent capacity of two inches of water. And neglecting short circuiting channelizing and immediate through-flow in a through-lift, we don't expect to have major quantities of leachate generated. That is, we don't expect to reach field capacity in our fill within the first 10 years because the refuse will be coming in as a blotter material faster than the precipitation is soaking into it. We definitely will reach a point that we're going to have an excess because we are in the humid east and there's some 30 to 40 inches of rainfall. It's at the planned point that we have to treat quantities of leachate that we'll have by circulation lowered the concentrations, that the treatment will be minimal in requirement for discharge. We'll then have to treat and discharge.

Ken Childs: Mr. Apgar, in your opinion, were there any interactions between the leachate and the matrix that the leachate moved through?

Michael Apgar: Yes.

Ken Childs: If so, what was it? The second question is frequently we assume that contaminants move out as some sort of a mass or plume of waste. Did your studies or results suggest that was or was not the case, or did you actually see bifurcating plumes of individual constituents? And third, what sort of a monitor system or how would you suggest a ground-water monitoring system be designed for a new landfill, one that's intended to detect contaminants rather than to pacify or satisfy regulation?

Michael Apgar: There is considerable interaction between the contaminants in the leachate and the earth materials that this leachate has moved through—that is, the strength of the leachate that we're collecting from the recovery wells in the aquifer is not nearly as bad as it is originally in the landfill itself. The sample chemical oxygen demand that we're pumping now at a rate of 2,000,000 gallons a day is down to somewhere around 25 mg/l. Granted, that's diluted with a lot of fresh water in the aquifer.

Ken Childs: Exclude dilution and talk if you can about absorption processes.

Michael Apgar: As the leachate moves through the ground, you filter some out and you have precipitation and absorption reactions. You have a whole host of things going that will decrease the concentration of the organics, inorganics. In the way of metals, we don't see any concentrations of heavy metals other than iron and magnesium oxide. So other metals, although they're there, and although the Redox conditions are appropriate or are such that they could be mobile, if they were there, they just don't move. I think that there is a lot of treatment or renovation as the stuff moves through the ground. It's just that we're not getting enough here before the leachate gets into an aquifer system. And once it's flowing through quartz sands in a confined system where there's no input or very little input of oxygenated fresh water, not much is happening and not enough is happening to decrease the concentration of the organics and the iron and ammonia before it gets to the wells and that's why we're involved in a contaminant recovery system there. Now, secondly, as far as whether the contaminants are moving in a plume or a slug, the geology is very complex. There's an interbedded sequence of sands, silts and clays, and in some places immediately adjacent to the landfill you can drill a well and the water will be fresh and uncontaminated as leachate has never moved to those positions. In other wells immediately adjacent to the landfill we find that the water quality is very degraded at times and it's pristine at others. This indicates that we are getting contaminants moving through the ground in pulses, even though the base of the landfill is continually saturated. Somehow we have slugs moving through and along one part of the flow path. We're seeing contamination at one point in time and no contamination at others. And this can be observed very strikingly by pumping a well. I have seen wells pumping clear water all of a sudden become very discolored and foamy, or wells that were producing foamy colored water clear up instantly.

So we have both things going. We have plumes and we have slugs here. We've gone to the philosophy that we shouldn't introduce leachate into the ground-water system. That's the concept behind all these liners. So as an effective monitoring program, first of all we ought to be monitoring what's going on beneath the base of the liner, if in fact we've got a liner. I think that to adequately monitor whether water-quality degradation is occurring in such a complex site is virtually impossible on a practical basis because we've got a hundred wells along Llangollen. We had to throw in 20, I think, before we could even get together a picture as to where contamination was occurring and where it wasn't. The first 4 or 5 wells right next to the landfill that should have shown contamination came up with good quality water. I think that what we do when we permit a landfill site to go in is to make the decision that the aquifer materials or whatever geologic materials containing water are near the site can be written off or degraded to an extent. That is, we're not proposing that we locate a landfill next to a major aquifer or on top of a major aquifer. I think to do that would be irresponsible.

Ken Childs: Can I conclude from what you've said that you could not predict with any reliability where a waste plume would occur downgradient from your landfill? And if that's the case, then a simple monitor screen of one well in the contaminant direction would not be adequate.

Michael Apgar: Well, I don't think that one monitor well downgradient from any waste source is really going to be adequate. Even if it's just one continuous sand layer 50 feet thick, you're going to pick up different water-quality results. I guess what you really want to know as far as monitoring is not how bad things are getting but just are they getting worse, is there leachate there. On a first cut monitor basis and if the leachate is there, then you've got to make a decision as to whether you have a more intensive monitoring or restrict water users from using the water.

Ken Childs: I have been involved with contamination and recovery studies and I know the number involving petroleum products where there's a residual that remains in the saturated zone indefinitely. Does this same sort of residual remain in the saturated zone for leachate? In other words, you can pump all the water out a hundred percent and then as new water comes in, there's still a residual that comes out and continually contaminates the new waters that pass through.

Michael Apgar: Yes, I certainly think that's the case.

W. B. Wilkinson: It has been suggested in Europe that perhaps domestic waste, domestic landfills, may be sited on aquifers on a thick and saturated zone and that the unsaturated zone can do the job of cleaning up the leachate. Would anyone subscribe to this idea? And if so, are there any simple guidelines they could suggest as to thickness of the unsaturated zone?

John Moser, Pennsylvania Dept. of Environmental Resources: We use a ratio of one foot of soil for every foot of refuse that you're going to place there. That is, put all that soil in before you put any refuse. We have to plan ahead how much fill you're going to put in there. This ratio was developed on the basis of some studies done at Penn State and it's not really entirely defined as to whether that's adequate in all cases or not, but we have used this and we are running out of sites very rapidly where we think this is suitable. It takes a fairly decent soil.

Randy Sweet: I think the most important thing is to be able to take a close look at the flow system, at the geometry of the flow system, and be able to predict where leachate is going to migrate.

Unknown: I think Mr. Apgar has done an excellent job in cleaning the landfill which was designed with some consultants 20-30 years ago. But why can't you put a well in the landfill itself somewhere, pump it out from the landfill itself and treat this by osmosis?

Michael Apgar: The option has been investigated. It's very difficult to get water to flow to any central point in this landfill. We've got 23 wells and five sumps that were dug or bored down into the landfill and we tried to investigate the rate at which we could get water to flow to any of these central points. We decided that we'd have to have a drain network with a spacing of somewhere between 100-200 feet in the trash to start collecting water out of the trash. To install this through 30 feet of trash, 20 feet or 10 feet of which would be saturated, would cost a couple million dollars. And since we weren't sure that was the way we wanted to go and the reverse osmosis studies show we have a large reject volume of water, we have about a 50% reject through the reverse osmosis system. 50% of the water came out good and we have 50% left, and it was just twice as bad as it was before and we had nothing to do with it. So we didn't go to reverse osmosis and collecting from the landfill.

David Miller: Thank you all for participating. We're supposed to reconvene across the hall.

Bull Session on Monitoring the Flow of Polluted Ground Water, and Artificial Recharge as a Solution to Pollution

Session Moderator: Wayne Pettyjohn, The Ohio State University, Columbus, Ohio.

Wayne Pettyjohn, Moderator: Our panel consists of these speakers: Bruce Labno, Pfeifer and Shultz/HDR, Inc., Minneapolis, Minnesota; N. Thomas Sheahan, Brown and Caldwell, Pasadena, California; Dennis Williams, Agro-Water Consulting Engineers, Tehran, Iran; Harry Nightingale, USDA, Fresno, California; and Stanley N. Davis, University of Arizona, Tucson. I'll now open the session for questions and discussion.

Gerald Hendricks, Sieco, Inc., Columbus, Indiana: I realize the value of a tracer, such as Freon 11, in both tracing ground-water flow, and in analyzing possible pollution. I have heard that the Freon 11 is, more or less, a common part of our atmosphere, and that perhaps the minor amounts that you might find in sewage effluent would not be revealing. I would like us to discuss the pros and cons of it or any other suggested better tracer for sewage effluents?

Stanley N. Davis: We do not as yet have enough actual data on Freon 11 concentrations in sewage effluent. We have sampled rivers downstream from industrial complexes where we assume that the rivers have received industrial waste, and these rivers contain Freon concentrations far above the concentrations that you would have in equilibrium with the atmosphere. Also, a well near a large landfill area in Indianapolis was sampled and very large concentrations of Freon were obtained from that well, suggesting that Freon might be indicative of some types of industrial effluent, plus landfill leachate. Theoretically, the problem with normal sewage effluent might be the fact that there's a tendency to aerate the sewage in some of the sewage plants and also the sewage contains a moderate amount of organic material both in suspension and in solution that might tend to absorb or actually combine with the Freon some way to take it out of solution so that it is not easily detected in the normal analysis. So my thinking at present is that Freon 11 may be useful in distinguishing between, say, sewage effluent and some industrial effluent, as well as landfill leachate, but it may not be possible to distinguish between sewage effluent and normal recharge water from the atmosphere. My coauthors are here. Perhaps they would wish to add something.

Thomas R. Schultz, University of Arizona, Tucson: One thing that maybe wasn't brought out in the talk that we're attempting to stress with this technique is the fact that it is a field-operable system and can be operated by people with very little training. You have the ability of taking more data right in the field and as was alluded to, it's very similar to the use of tritium sampling, which is quite expensive. So if this technique should become valid, many small consulting firms and public agencies could use it.

Gerald Hendricks: I infer from what you're saying that a gas chromatograph would be used as an analysis and if so, other people could assist in obtaining the data rather than a specialized group of high-quality lab people?

Thomas R. Schultz: That's right. Many other environmental tracers have many different origins, such as tritium and chloride. It's difficult to really know what the sources are. With the Freon, we know exactly how and when it was first introduced into the atmosphere so it should be much easier for people to interpret the data. In regard to your question on errors, I think, although I have no data to demonstrate this, neither ourselves nor Glen Thompson of Indiana University has gotten to the point yet where we could look at analytical errors in the field. One of the projects that we're getting ready to undertake is to calibrate or verify this technique in the Edward Limestone in Texas. There's a model that exists for the Edward Limestone which takes into account the flow and also mixing and dispersion of tritium. We would like to verify the Freon technique with this model, and in the process, we would be able to get some hard data to answer a question such as yours.

Stanley N. Davis: To amplify the questions you brought up concerning errors—one advantage in field analysis, of course, is that you can repeat your sampling on-site and make sure that you're getting consistent results from the gas chromatographic system. Furthermore, at the present time, Glen Thompson is preparing and will, I assume, distribute some standard samples so that calibration can be made between various workers interested in low values of Freon concentration, so we not only can field-check the sampling, and hopefully reduce any errors that may be present in the sampling technique, but we can also assure ourselves that we're getting some reproducible results in terms of other labs.

Kenneth W. Lustig, Panhandle Health District, Coeur d'Alene, Idaho: As a small health department, we historically use rhodamine dye for tracing septic tank effluent into adjacent lakes. We're presently involved in a 208 effort with a couple of agencies on an aquifer study. It's been proposed, in order to try to substantiate that the effluent from the septic tanks is recharging the aquifer, that we put some kind of tracer in it. It's been suggested that we use rhodamine. Would someone care to comment on the advantages or disadvantages, outside of tritium, of something that a small health department could use?

Thomas R. Schultz: Rhodamine is quickly absorbed to clay particles. If you have an aquifer containing a significant amount of clay, rhodamine or any of the fluorescein dyes would be an extremely poor choice. Chlorofluorocarbons have the lowest surface energies of any organic, so they're less likely to be absorbed by other organics or clays. You need something that could be detected at extremely low concentrations, could be demonstrated not to occur in the environment and might be the sort of thing that would hold up in a court of law. Also it's possible that if there were more than one polluter, each polluter would be obligated to inject a small quantity of a certain tracer into his effluent and these can be separated easily. This would cost about $2,00 to $3,000 in parts plus labor. The operation of the equipment is much like operating a component stereo system, almost anyone can do it, and the analytical techniques are firmly established.

Stanley N. Davis: One possibility of a cheap tracer might be bromide. It's not very concentrated in the environment, it is relatively nontoxic, at least most of the bromides, and if one has a specific ion electrode to pick up bromides, you can usually measure down in a few tenths of a ppm. It depends on the amount of dye solution that you're going to anticipate, whether or not this would be practical, but bromides have been used on some occasions.

Kenneth W. Lustig: The situation we're in is where we've got an area pumping 5,000 gpm with less than half a foot of drawdown. The surrounding area is being subdivided. We have a 20-acre tract dedicated to us by a small industry, and we dug a test well to sample the ground water with a tracer. We're involved in a lawsuit at this time with the building industry. We're at a very crucial point and we want to use something that is going to stand up in a court of law. The way the whole area develops is going to depend on what happens in the next year or two. We have about a year or year and a half's worth of samples now and the only thing that's showing up that we can really count on are the nitrates, which peak under the subdivision. We've got less than a ppm elsewhere, and 9½ under the subdivisions.

Stanley N. Davis: As you describe the problem, I would doubt whether bromide would be a good solution because you would probably have to use very large quantities and there might be some public health question.

Kenneth W. Lustig: What constitutes a large amount of clay? We've got areas where we mine the gravel and never even wash it, but the well drillers we talk to say, "We ran into two or three inches mixed with clay." Would that be enough to filter some tracers out?

Thomas R. Schultz: I don't recall exact data but I would think that would be enough. It would at least reduce it to the point where the hydrogeologists from the other side could make the data questionable.

Gerald Hendricks: I would ask Mr. Schultz if the Freon equipment setup is going to be published?

Thomas R. Schultz: I believe Glen Thompson will probably be describing some of the analytical techniques in a publication he's working on now. It should be within six months or a year that a person with proper facilities could build one of these devices himself.

George Hoffman: I have a question for Dennis Williams. Did you consider increasing the number of pumping wells and doing away with the well barrier?

Dennis Williams: One of the purposes of the project was to illustrate the principle of conservation of ground water through underground storage. Possibly we could have made up the difference that we got by recovering previously stored water or by importing additional water, but that would have defeated the purpose of the project. There are many areas in which we'll have to use this principle of bringing in excess water during the winter season so that we can store it underground either through injection wells or artificial recharge basins and then recover it during the irrigation season.

The hydrologic situation as it exists without any extraction or recharge is in equilibrium; in other words, there's fresh water in certain areas, and our project was to superimpose a ground-water mound upon this and then recover the mound. The salt-water ridge is more or less stabilized now through natural conditions. We're only pumping out the mound. In other words, we can't start the project pumping first and then recharging. We built the mound about eight meters high. By the time irrigation is to begin, and then durring the irrigation season, our recovery wells will degrade this mound down to the original level before we started.

George Hoffman: How did the salt-water aquifer get such great heads? They're way above sea level and way above the head of fresh water?

Dennis Williams: The only explanation we have is this zone of high pressure fossil waters, laying below 100 or 200 meters, was formed during the transgression and regression of the ancestral Caspian Sea in Plio-Pleistocene times. The overburden was building up, and as a result the pressures were increasing and a lot of water was squeezed out of the clays which compacted more than sand, and the water then migrated into the sand zones, with the result of increasing pressures. Our exploration has shown that there is no hydrologic continuity between this lower zone and the fresh circulating ground water due to an aquiclude, 20 to 30 meters thick, consisting of very tight organic clays with some silts. This zone, for some reason, is just sitting down there and it's geologically sealed off, but when you do tap it, the water comes out of it like sea water.

George Hoffman: Then you don't think it's connected to the sea? It must be sealed off?

Dennis Williams: It may be connected to the sea, but our project is located 15 or 20 km inwards so we really haven't studied the continuity. The aquifer is below sea level but it may outcrop somewhere.

George Hoffman: Are these heads typical in other parts of the country or the world?

Dennis Williams: My experience in California is we don't have any heads near that high. For example, in Owens Valley, where I've done a lot of work, we've encountered maybe one atmosphere a head. We've encountered these tremendous heads in several areas in Iran, not only along the Caspian Sea but also along the Persian Gulf area. But I'm sure it occurs when you have buildup of sediment with a lot of clays.

N. Thomas Sheahan: Is there much of a temperature increase in this lower salt zone? Could high pressures be geothermal?

Dennis Williams: No, we didn't observe any temperature increase. The temperature of the lower water was the normal ground-water temperature; however, some of the shallower wells, which smelled of hydrogen sulfide gas had temperatures close to 80 degrees F; however, the water quality was 700 or 800 micromhos/cm.

Gerald Hendricks: Would Mr. Sheahan describe in more detail the type of wells that they used for their project and why they used those wells?

N. Thomas Sheahan: We had some principal points of interest in the design criteria for these wells. First of all, knowing that we were going to be recharging reclaimed water and the potential for suspended solids which also goes with it, a potential for plugging of wells, we do expect to have a maintenance problem on the injection wells, because of suspended solids, and as a result the materials of construction and the types of construction were designed with this in mind. Also, there's a serious research program that is starting and will be going concurrently with operation of this system for a number of years, looking at water quality aspects of injection and extraction of the reclaimed water into underground formations. Therefore construction materials were selected which would have the least possible effect on water quality, so that our water-quality variations would be minimized and directly related to the geology and hydrology. The third thing was the physical requirements of injection wells, somewhat higher pressures and structural problems. As a result of these three major areas, we selected Schedule 80 polyvinylchloride casing. We selected an 8-inch well size for the injection wells principally for the requirements of pumping equipment. The well screens were 304 stainless steel wire-wound well screens which, in my opinion, is probably the best type of well screen for the most efficient operation. The stainless steel also was selected because of its nearly inert character and the least chance of affecting water quality in any way, yet providing sufficient structural strength in the screen area. The inside diameter of the screen is the same as the inside diameter of the casing, so with this type of configuration we can effectively use both mechanical and chemical development methods in the future. We can use various types of chemicals.

The wells are gravel-pack construction. The importance of this is to provide one additional design buffer between the well screen, the last point that you have the control of injection water, and the formation phase, where you lose complete control of the injection water. By using a gravel-packed well, we can install a gravel pack that provides an effective filter as well as a medium to hold back, and so by designing the gravel pack in this way, sediment that might be deposited in the well during injection would be filtered out near the well face, we can more effectively remove these with a maintenance program in the future. The gravel-pack was a 10 or 20 type gravel, nearly 1.0 uniformity coefficient. Each of the wells is double cased and double screened. The deep well is 8 inches in diameter, and the smaller well is 6 inches.

Gerald Hendricks: What was the diameter of the gravel pack, and did you vary the anticipated rate of application in any one well depending on the formation and size?

N. Thomas Sheahan: We will be varying the rates because there will be variations in permeability, aquifer thickness, and geology in the area.

E. W. Ramsey, Virginia State Water Control Board, Richmond: With respect to the quality of water and its injection into an aquifer, have there appeared any problems with respect to pressure and also with respect to compatibility of the injection water and the aquifer into which the water is injected?

Dennis Williams: The injection pressures that we're using are designed not to exceed two atmospheres, which is 28 or 30 p.s.i.

Harry Nightingale: While on this subject of recharged wells, we have one at our Leaky Acres Recharge Facility. At this facility we're using about the top five feet of soil to filter the canal water. The water is collected and it is gravitied down the recharge well. This recharge well is a cheap well. Anytime you start a recharge operation, you are going to get clogging of the interface between the well and the aquifer. The system which we are working on is just about the opposite of a good water well; that is, the well is a sander when you pump it. The well is built with a four-foot diameter conduit pipe down to the first 20 feet. An eight-inch well goes down to about 250 feet, but the well is gravel packed with 1-1½ inches of nice round river rock.

This year we started out at about 500 gpm—this is a gravity-fed well—down to about 350 gpm. We've had some clogging on the surface of the aquifer interface. We don't pump it very hard. We want to mine the sand that's in the aquifer. This is where the colloids have been filtered out. We mine this sand into the well and pump some of it out. When you do this, the gravel around the casing starts moving down and then out into the aquifer. We replace the gravel at the top and this way we hope some of the colloids get filtered out. We can measure the colloids and actually recover the original recharge capacity. At this facility, the ground-water mound beneath the basins is pretty well in a chemical equilibrium with the canal water and the colloids in movement are in equilibrium with the soil water as well as the ground water. When these colloids are injected they are not going from a low saline to a high saline environment and continue to precipitate; they are going to an environment similar to where the colloids came from, so they move a tremendous distance. We know they've moved at least 400 meters out into the aquifer.

Wayne Pettyjohn: Would anyone attempt to answer the problems that we all face in trying to get technical ideas across to administrators, politicians and other people?

N. Thomas Sheahan: At the risk of maybe oversimplifying things, we ran an injection test, and one of the most important parts was a demonstration project. We did not, at that time, deal with the water-quality aspects of it, but perhaps a demonstration test over a period of time during which sufficient water-quality data is made available to draw some conclusions, however valid, and maybe this is the easiest way. Administrators are much more receptive to a comment like, "We have done this," rather than, "We can do this."

Wayne Pettyjohn: We've had several questions on construction details of recharge wells, and I'd like to briefly explain some that I constructed several years ago which were described in *Ground Water* about ten years ago. This was a situation where there was sand from the surface down to a depth of about ten feet, about fifteen to twenty feet of silt and clay below that, and a major aquifer underlying that. Water levels were down about 70 feet. We constructed a Y-shaped canal system attached to a basin and pumped raw unfiltered water directly from a river, discharging it into the basin. When the basin was full, the water flowed down the Y-shaped canal system. There was a lot of recharge from the basin itself. In the canal system we constructed two types of wells. The first type were bored wells deep enough to go about ten feet below the bottom of the confining bed. They were 30 inches in diameter, filled with sewer rock and covered with pea gravel. As the highly turbid water flowed down the wells, they plugged quite rapidly and it was almost impossible to clean them. Using a dragline with a ¾-yard clam attachment, we excavated 4 holes, which are, in reality, about 12 feet in diameter. About 20 feet of corrugated culvert, 6 feet in diameter, were installed and gravel-packed with pea gravel on the inside and the outside. Obviously, the rate of recharge through the large-diameter wells, which were very inexpensive, was high. Of course, with time, the rates decreased as the sediment began to accumulate in the gravel pack. It was a relatively simple matter, however, to remove the gravel pack with a clam, which was then stockpiled adjacent to the recharge site and clean gravel was installed. After a year or so, much of the gravel that had been removed was washed clean of the sediment that had accumulated. It could then be reused. It was a very inexpensive method and rather primitive, but even so, we were able to recharge about 4,000,000 gpd, although this decreased with time. The original proposed solution to the water shortage was a pipeline to the Missouri River; 43 miles of pipeline at a 1959 estimated cost of $12 million. The city had neither the money nor the time to construct a pipeline and that's why the recharge system was built. The total system, including land acquisition and the construction of an $87,000 dam, was about $192,000. Furthermore, when the system was put into operation, the water level rose more than 22 feet within 6 months of operation. The point I'm trying to make is that it is possible to develop artificial recharge systems quickly and with limited funds.

I'd like to ask a question of Mr. Nightingale. I noticed on one of the slides that at the beginning of each recharge period the mineral content, whether it was chloride or nitrate, increased and then it declined rapidly. This appeared to be cyclic. Would you comment on this?

Harry Nightingale: Well, say we've been recharging all summer long and we've reached a low value EC for that year. When the water is turned off in October, the profile drains rather rapidly but, when the profile is pretty well drained, there are zones of soddy clay soils which are micropores, and we feel there is diffusion of salt from the micropores into the main flow channels. There appears to be some downward percolation of this more saline water, but most of the water moves out to the large openings so when the next recharge period occurs, the water goes through the bigger channels rather than the smaller ones since the salt has diffused into these channels, and a new peak of salt is leached into the ground water, and then, as the recharge period continues, the salinity continues to drop down.

Wayne Pettyjohn: Might the increase represent water-soluble products that are forming in the unsaturated zone between recharge periods and you're just merely flushing those out?

Harry Nightingale: Dissolution of the rocks does occur as well as evaporation from the soil surface. We've looked at the EC of saturation extract after recharge and found it is very low, but after 3 months, in the wintertime, it will be back up. Some water must be moving back up, evaporating, and the rocks are slowly dissolving. I might add, since you were talking about recharge shafts, that we've done considerable work on recharge shafts, and one we had set up this summer ran for 66 days. We put 6.4 million gallons of water down this one shaft. This is irrigation canal water, but it does go through one screen filter, about like a flyscreen, which is just enough to filter out any moss, beer cans, etc. Again, the recharge shafts are built different; they're 4 feet in diameter and 32 feet deep. We've got about 10 or 12 feet of good sand, but again we go to gravel which we can add as we mine the aquifer. The shaft is built so that we have the conduit pipe which is a 10-inch pipe, and at the bottom of the shaft there is an iron cage. On top of the cage there are some rocks. In this case as the recharge water flows out into the aquifer, it becomes plugged. On this shaft recharge is started at 475 gpm, but after 2 days, it was down to 100 gpm. After 66 days it was down to about 50 gpm. To increase the rate, the first thing we do is try to mine the sand. The top of the shaft is sealed and when a submersible pump is turned on, you get a shock connection and a high velocity. The sand sometimes will pump out but a lot tends to go down into the cage where we can get at it. Our idea is that these shafts are very cheap to build, they can be connected to an irrigation canal system and they only occupy a few square feet. But our experiences show that the canal water should be pretty clean. The water which we're using has a turbidity of about 3 to 5. Sooner or later the shaft will need to be abandoned. Only a few hundred dollars invested in it and all you have to do is move over about 15 feet.

Wayne Pettyjohn: It seems to me there's no question but that artificial recharge techniques will provide a solution to many of the water problems that we have. Why they aren't used more is certainly beyond me.

Stanley N. Davis: A short question for Mr. Nightingale. For many years, storm water in Fresno was drained into so-called dry wells. Are those still operating or have they abandoned them?

Harry Nightingale: Basically they've been abandoned.

Ken Brewster, Illinois Division of Water Resources: With a little background, I'd like to address this to Mr. Sheahan. In the Chicago area there's been a proposal to inject tertiary-treated sewage effluent that has also undergone advance treatment, into the two primary rock aquifer systems. Do you think you could have a doublet of the type of design that you proposed that would adequately remove a contaminant? What problems might be associated with such a project? The initial project, as proposed by the Federal EPA and the Metro Sanitary District of Greater Chicago, would require an extraction system during the test. If in fact it did work, the extraction well would be put on standby for emergency purposes only.

N. Thomas Sheahan: The key to the injection-extraction doublet operation, within a limiting streamline, is a regional gradient. Without a regional gradient, there is no limiting the streamline, so it has to be there naturally or created artificially by overpumping the extraction well. To determine what the water quality would be after injection and passage through the aquifer when it comes back out the well, the wastes could be mixed with a certain amount of native aquifer water because of the overpumping of the extraction well required to produce the artificial gradient. This may produce a problem. The only way around that would be to have sampling points between the two wells to use for basic data. The extraction wells could be used simply as a tool to make the system work and completely extract the water. On that basis, I would guess, without knowing any more about the geology, that, in either aquifer, you could construct such a system and adequately control all the water injected by use of extraction wells. When you talk about injecting into two different zones, though, this may unnecessarily complicate the hydrology and it may or may not be possible to extract all the water from multiple-screened wells.

Wayne Pettyjohn: Ladies and gentlemen, I'm afraid that our time is about up. We are supposed to reconvene across the hall for a summary session. Thank you.

Bull Session on Managing the Movement of Contaminants, and Protecting Mines, Wells, and Pits

Session Moderator: Tyler E. Gass, NWWA, Worthington, Ohio.

Tyler E. Gass, Moderator: Our panel consists of these speakers: John Fryberger, Engineering Enterprises, Inc., Norman, Oklahoma; Otto Helweg, University of California, Davis; Elmer E. Jones, Agricultural Research Service, Beltsville, Maryland; and David Walz, Virginia State Water Control Board, Richmond, Virginia.

Fred Hoffman, U.S.EPA, San Francisco, California: I would like to address my questions to Otto Helweg. This morning he discussed the use of ground water in an agricultural community where the water was becoming more saline downgradient. In many places, we have similar situations. I'm thinking of an example where we have a valley such as you described this morning with increasing downgradient salinity. Artificial recharge has been established upstream with water that has higher TDS than the water that is in place now in the upper parts of the valley. Have you thought how ASTRAN might work, or how you might manage that type of situation?

Otto Helweg: Obviously to make the system work you've got to have a water quality gradient someplace. Ideally if you could start upstream with high-quality, usually surface water that's higher quality than the ground water, then you've got it made. If you don't have that, the only other thing you could do would be to maybe set aside a section of the upstream aquifer that is of high quality or the highest you have and transfer that downstream and depending on your setup, the imported water then could be recharged at some point where the ground-water quality could be perhaps just a little bit worse than the recharged water, but I realize that you might need it upstream rather than downstream, so it's a little difficult to comment.

Fred Hoffman: Okay, just in general then, your ASTRAN is only going to work if you have a perfect situation: good water quality upgradient, increasing salinity downgradient?

Otto Helweg: I think it's more general than that. I have never run into a river basin that you did not have this water-quality degradation as you go downstream. If you have that, the method can work, technically. The only constraint might be the political, institutional or legal constraints whereby if there was no way you could use the highest quality upstream water downstream.

L. Jan Turk, University of Texas, Austin: I can show you a lot of basins in Texas in which the opposite is true. Our streams flow from the arid to the humid regions and the water quality improves toward the coast so we do have those situations there. These are major river basins. They flow from the western arid region to the coast and the rainfall dramatically increases. We have a lot of salt springs; they start out as saline rivers, and there's a tendency for those that flow across the northern and central part of the State to improve in water quality. Some of them get more polluted from bacterial sources.

Otto Helweg: I would say the ASTRAN method would be of no value. On the upstream part where it is arid, do you see a reversal in the gradient?

L. Jan Turk: Generally speaking there's a fairly steady improvement downstream because of tributaries coming in, and the increase in rainfall and increase in runoff results in dilution of the water as you go. This is really unusual because most streams flow from the humid areas to the dry areas and this is just exactly the opposite.

Tyler Gass: I think the situation would become more common as you approach more and more humid areas even if the stream started in the humid areas. There are many streams in the eastern part of the U.S. where there are natural forms of contamination that do become diluted out as the stream goes downgradient.

George Robinson, Peabody Coal Company: We have a situation in the upper Colorado River Basin where they have the salinity control plant. We wanted to dewater a mine which had fairly high quality water to add to the system, and applied for an MPDS permit and were originally or initially declined because the fact is we are adding the TDS to the system and regardless of how much water we're adding to it they stated that the water would be evaporated and the TDS would be still in the system so they declined us the permit. Now we have no alternative but to close the mine down even though we are releasing higher quality water to the system. I would like to see your management plan adopted by EPA and I think some of the regulatory agencies should be aware of the systems we're talking about because with these constraints placed on no discharge, it makes it almost impossible for industry to work within the guidelines of the law.

Elmer Jones: In Public Law 92.5 it basically states, I believe, that a natural discharge is not a pollutant; if man controls that discharge it then becomes a point discharge and is a pollutant. Have they effected that practice of trying to pump the salt water into storage ponds and discharge it at high-water flows?

John Fryberger: To my knowledge the Corps of Engineers has not chosen that as a potential solution and the reason is during high-water flows the rivers carry a great load. The concentration is lower then but the total load in terms of tons per day is very, very high. This is primarily because of the re-solution of the salt that has been deposited in the salt plains. This salt is redissolved and carried out by the flood waters so although the concentration is not as high, still the total load is very high during the flood stage. The Army Corps of Engineers have considered building a low-level dam across the stream, just downstream from the source of the salt and pump from the reservoir behind this dam into the off-stream evapotranspiration sites. This technique for control has been evaluated and in some cases it will work all right but in many other cases it will not because most of the load, in terms of tons per day, is carried down by the flood waters, so to capture the water and then release it into the flood waters is already being done under the natural system. Salt is being captured by evapotranspiration forces, being held in place and then released to the flood waters.

Elmer Jones: Isn't the salinity in low base flow the greatest concern?

John Fryberger: No, I would say it's the flood-water load that is of greatest concern. For instance, the Red River is dammed at Lake Texoma on the eastern side of the border between Oklahoma and Texas. Lake Texoma originally was intended to be used as a water-supply reservoir for some of the towns in northern Texas. It can't be used because it's too salty and getting saltier, and it's getting so primarily because of the flood flow that picks up all that salt.

Lee Burton, Oklahoma State Department of Health, Oklahoma City: On base flows, normally in those streams in Oklahoma or in that part of the country, the base flow is usually made up of contributions by return flow from the terrace deposits and alluvial material in the local area and usually it's of much better quality than that of flood flows. The flood flows do pick up the salt, and therefore it's during floodtime they are of particular importance as far as the salt load. When they do have flows, in your equation you presented today, does it take into consideration the period of time when those streams sometimes will have a loss of flow at one point and then later on downstream they will have an increase in flow? I know you were developing formulas so you'd have a general secretion of water downstream but some of those streams, particularly through the salt zones, do have a loss of flow.

John Fryberger: Yes, that's very true. In the upper reaches in most of those rivers there are long periods of time when there is very little if any surface flow and all of the water that is in the alluvium and coming into the alluvium, say, from the ground-water inflow into the alluvium is being lost by evapotranspiration. In fact, the sum of the spring inflow and the natural ground-water inflow below the spring is being consumed and lost by evapotranspiration so the only time when you have base flow, really, is following a flood event. The flood saturates the alluvium above the water table and it forces the water that's in the alluvium down and up into the river channel and comes out as base flow. The base flow we're talking about in the equation in some reaches of the river is only occurring following a flood event, but it is the base flow that carries the salt out of the alluvium. It's the only mechanism you can have to get the salt out of the alluvium. Sometimes those streams are dry or there might be little ponds here or there but they're quite dry.

For instance, the amount of ground-water inflow per year in Reach 1 of the Red River we calculated to be 59 acre-feet per year whereas the evapotranspiration losses for that same one-mile stretch were calculated to be 844 acre-feet per year so, in some of these rivers, you have a lot more evapotranspiration loss than you have ground-water inflow. Of course, in addition to the ground-water inflow we have precipitation which we calculated to be 428 acre-feet per year; and we have our flood inflow which was 449 acre-feet per year, so the sum of all the inflows is greater than the evapotranspiration loss. The difference between those two is the base flow.

Don Lundy, University of Wyoming, Laramie: When you write QC_a, did that mean Q times the concentration or does that mean just the volume?

John Fryberger: No, that is Q times the concentration. In each factor you have a quantity of water times the concentration of that water.

Don Lundy: I see, but when you get to precipitation in evapotranspiration, you don't have a concentration following the P or E and you have expressed those as acre-feet so —

John Fryberger: They could well have a concentration. It is explained in the paper and I should have explained it today, the concentration for the P and the E values is assumed to be one. You could have C_e in all the concentrations of the evapotranspiration losses and that is assumed to be one for both P and E to have continuity.

Otto Helweg: Isn't there any possible way to come up with a reservoir release policy downstream that could, say, pass on the flood flows and maybe maintain the low flows? In other words, try to store low chloride water and pass on the degraded or salty water?

John Fryberger: The chloride concentration during a low-flow period is higher than the concentration during the flood periods but the total load in tons per day is lower during the low-flow periods, so you would want to store the water with the lowest concentration which would be the flood water. But still, the flood waters are pretty high in salt and they're carrying a high load in terms of tons per day.

Otto Helweg: I don't understand why the reservoir is degrading so badly; in other words, what was the water quality when they first built it and how come it's getting worse now?

John Fryberger: It's degrading because of the added evapotranspiration losses and concentrations by evapotranspiration losses at the reservoir site itself. You're getting water in all the time but you're losing water from the reservoir all the time, too by evapotranspiration.

Elmer Jones: When you consider this on an annual basis, the number of low-flow days exceeds the number of high-flow days, so on a long-term basis what is the ratio of salt transport?

John Fryberger: I can't answer that directly; I might approach it though indirectly. Not for this flushout study but for another part of the study, we did evaluate the effect of a low-level dam. It was desired to prevent 80% of the salt from going further downstream. The height of the dam to do that and the amount of water that they would have to pump out from behind the dam, especially in this one river, was prohibitive because of the greater load carried by the high flows. I don't have a ratio of the total load carried by high flows versus total load carried by low flows because it varies quite a bit from river to river. In some of the rivers most of the load was carried during low flow, and in those rivers a low-level dam would be effective. In other rivers most of the load was carried by the high flows, and in those cases the low-level dam would not be effective because you'd have to have too big a dam.

Tyler Gass: Yesterday, Wayne Pettyjohn came up with his drip theory in which ground-water contamination—in his case, saline solutions seeped out of evaporation ponds for oil-field brines. According to him, you would get a massive drip or a contaminant pulse entering the ground-water system and then moving out to the stream. In arid areas, you have a situation where you seem to be getting a pulse-type effect due to periodic high runoff or floods. How would this philosophy of taking data affect some of the projects that you gentlemen may have worked on?

John Fryberger: I can give one opinion, anyway. As far as this drip theory would apply to the flushout equation that I've been talking about, I really don't think it would apply. The flushout theory is intended to be applied over long periods of time. The time period for the one-mile travel in the river that I had described is pretty long, 15 to 20 years and his pulse idea wouldn't come into play. But we fully recognize, for instance, following a flood event you would have a slug of water that goes down into the alluvium and mixes with the ground water. That slug of water is moving through the ground water and finally is discharged as base flow so if ground-water quality is measured at different places in the alluvium, you'll be measuring the effects of these slugs. If you were to measure the ground-water quality at one place in the alluvium and say, "This is characteristic of all the alluvium in this cross section," you wouldn't have a very valid number. As far as determining the water quality at one point downstream in the alluvium you have to measure the water quality at several points in order to negate the effect of the drip theory.

George Robinson: In Ohio I agree that the drip theory applies as to what Wayne was mentioning yesterday, but in Montana where we have a detailed reclamation program going with moisture monitoring using neutron probes, the drip theory, at least after two years, doesn't apply because we haven't had a moisture front really pass to any great depth. The moisture front from two years of data has only penetrated about a maximum of 10 feet. The rainfall here is about 8-15 inches.

Donald Keech, Michigan Department of Public Health, Lansing: Mr. Jones, was the air testing done below the casing level in the lime rock?

Elmer Jones: Yes.

Donald Keech: Specifically what mechanism did you use for the air test device? How could you pinpoint the exact level of where either water was coming in or going out?

Elmer Jones: Your casing is set in rock such that you can depress water 10 feet below the bottom of the casing with air and maintain air pressure at that point; you have a sound grout seal. In our tests we had four correlations for this but only the pumping tests where we took samples under very carefully timed conditions and the air pressure testing could be directly related to the adequacy of the sanitary protection of the formation. A caliper log at this 79-foot level revealed a slight enlargement of the bore but you find in open bores, many places where you'll have a slight enlargement. That means nothing as far as sanitation protection is concerned. The natural gamma log indicated a higher percentage of clay at that depth. In dry weather, we would pump the well and take a sample at one minute, two minutes, four minutes and on a timed schedule like this. At the two-minute pumping time we might get a jump of 100 times in our coliform count, and then by the time you were drawing water from the bottom of the bore, this would have dropped down to less than 10% of what this peak was. This happens to be the time it required water to move from this depth up to the pump and into our sampling point. The air pressure testing, to evaluate it quantitatively, you have to convert your air from volume to pounds because the density per cubic foot changes constantly as you depress the water further in the well. But we find that when we convert to pounds, which we can then convert to volume at standard atmospheric conditions, a point like this will have a fairly constant air loss with time. After open bore grouting, this is to 100 feet, the well was air tight at 35 pounds pressure to 100 feet which was 40 feet below the bottom of the casing.

Donald Keech: Do you have charts on that, Elmer, that are available?

Elmer Jones: No. I wanted to get a publication out on this but at this time we don't have experience with certain formations and only in wells where the casing supposedly terminates in solid rock can we tell you how to have confidence in it. We use this on testing wells in a monitoring area and here the wells are screened with a fairly coarse sand. As the water is pushed down into this screen it will drain out and then be displaced by air, and your rate of air loss will increase. However, we have been able to identify wells that we decided obviously were not constructed to specifications. Where they were supposed to have 10 feet of grout, they had about 10 inches. To go out and use this method to evaluate a commercial producing well that would be cased all the way down and screened into sand, I would say that you could only use it with confidence within the casing. You have an air leak in the casing or near your upper casing installation if you cannot maintain air pressure above the water table. We found in this situation that if we cannot maintain 10 feet of air pressure in the casing—if we take a suction sample from the top 6 inches of water in the well, our coliform count may be several hundred times that of the pumped water sample, which is a very clear indication that your contaminant is entering above the water table and the concentration at the water table in the well is the highest.

Lee Burton: You're speaking of this contamination that's actually coming from the ground water, but is there any possibility that when you're developing your well and the use of your grout material that you have induced a contamination there, and that you're just getting a return from that rather than from the ground water itself?

Elmer Jones: No, we're talking about fecal strep, fecal coliform, total coliform. The high pH of the grout will disinfect the well for a period of time because the pH of your water immediately after your grouting operation will be around 12. After a rain the coliform count will go up into the 10,000 or 100,000 area and then when the water shifts to this discharging mode, as the water table continues to rise, you drop off to zero or to a much lower count, so the contaminant is not in our construction.

Donald Keech: Elmer, on the air test did you measure the water level at the same time you were pressurizing the well?

Elmer Jones: We measured the water level before and after, but not during. We use the initial water level plus the pressure of the air converted to altitude, feet of water to determine what the water level is in the well and in the event that during the testing period there was a significant change, we also measure this afterwards. Normally the water level will recover to essentially the same elevation.

Ed Ross, Minnesota Department of Health: Mr. Jones, you mentioned something about a cementing shoe. I would like you to explain how you grout these wells, and how you were able to find a zone that you didn't get a complete grout. How did you locate that particular zone?

Elmer Jones: In the sinking shoe grouting technique you open your bore to the depth necessary to obtain a productive well and in doing this you should watch your formation to find the desirable depth to terminate your casing. Once you are sure you're going to have a productive well, you then ream the bore to the depth that you're going to set your casing. In the Johnson Well Manual you'll find this sinking shoe technique described but they recommend filling the well with clean plaster sand. We found we couldn't use this because we cannot buy clean enough plaster sand. So we back filled with No. 3 well gravel up to within about two feet of where our casing was going to terminate and then we put just about two gallons of fine sand in to separate the grout from this. A tremie is used to implace the grout. We placed approximately twice the theoretical volume of grout required to fill the annular space. The sinking shoe is either a drive shoe or a coupling in which you have set it in a funnel and made this pointed nose with a quick setting cement. This is screwed on the bottom of your casing and as you run this down the well the grout is displaced up the annular space between the casing and the borehole. We try to hold our sinking rate on the casing to about 2 feet per minute. Then each time we make up another joint of casing we give the casing 5 extra turns in the borehole so that we have a high radial velocity between the casing, the grout and the bore in addition to the high vertical velocity that we get when the casing is sinking so there is no possibility of having any bridging in this. This is why we have great confidence in this grouting technique. The point at which the contaminants were entering Well A was 19 feet below. When Well A was reconstructed it was cased and grouted to 60 feet and at 79 feet we had contaminants entering and were losing air in our air test. This was 10 feet below the bottom of the casing. When you have a sound casing and a partially grouted annular space and you have a contaminant entry point behind this, all your testing will tell you is that your air is leaving the well at the bottom of the casing and you can't tell at what point behind the casing it is leaving the bore.

Ed Ross: That explained the sinking shoe but maybe I didn't understand you about finding this one void space. How did you determine where that was?

Elmer Jones: As the air pressure was brought up in the bore, we keep a record of the time it takes for the pressure to increase. If you go past a point at which you're losing air but you're not losing air at the rate you're putting it in the bore, the pressure will continue to increase but at a lower rate and you may or may not reach a point at which you lose all your air. The test plug that we had been using was rated for 15 psi. We put C-clamps and angle irons on this to hold this down and would go on up to 35 lbs. pressure. I would be extremely reluctant to advise people to use these test plugs in excess of the manufacturer's recommendation unless you have taken great care to see that the test plug cannot blow out because it is heavy and should it hit somebody you could almost be sure to have serious injuries.

Donald Keech: Elmer, how did you grout the open bore hole?

Elmer Jones: Again we started with the same technique that we used for the sinking shoe. We back filled the bore to the depth at which we wanted to bring our grout down to. We placed our tremie down there to where our grout went to the bottom of this with the plastic barrier to separate the sand and the grout and ran the grout down the tremie and up, measuring, again, with this little hydrometer to see how much grout it was taking to fill. Actually, in this case, it was seeing how far 10 gallons of grout would rise in the bore and from this we knew how fast the grout was rising below 79 feet. It took more than 10 gallons to fill this at 79 feet. From there on up it took essentially the same gallons of grout per foot of bore as it did before, and then we brought the grout up into the casing about 10 feet, so that if there were any settling we would have continuity. However, had we stopped grouting at 70 feet this would have been adequate because the air test had previously shown that once air had bled out of the well to the 79-foot level we could hold air for an hour at that level with no discernible pressure change.

We redrilled the well. When the grout is at low strength, I would not be hesitant to take the grout out of the well and at that point I could do it with an air compressor.

Tyler Gass: Mr. Walz, to what degree was there any public opposition to putting the site where it was located?

David Walz: We went before the City Council and made a presentation as far as the information we had gotten with the site evaluation. The City really had no way of saying "No" to us because they had accepted the problem that had come with it. The City agreed to use part of their landfill site to dispose of the Kepone. We had no reaction from the residents.

Maxine S. Goad, New Mexico Environmental Improvement Agency, Santa Fe: Mr. Walz, who actually did the waste disposal operation?

David Walz: The actual disposal was carried out by a private contractor engaged by Allied Chemical Corporation but the disposal process and design was put together by the State—by the Ad Hoc Technical Committee with a representative of Allied Chemical in concert with us. At all times during the disposal there was a State Supervisor at both the plant and at the disposal site so it was done by private contractors but under State supervision.

Maxine Goad: Dr. Fryberger, if I understand it correctly in your flushing equation which had terms which were water quantity multiplied by concentrations, what you had were salt loading balances, is that correct?

John Fryberger: Yes.

Maxine Goad: May I assume that the salt lost through evapotranspiration is extremely small?

John Fryberger: Yes, it would be very small. The concentration of salt is assumed to be one for purposes of the equation. It's not even shown in the equation for either the evapotranspiration or the precipitation. In the paper it is explained that those values are assumed to be one.

Maxine Goad: Why did you choose to even include those terms in your salt loading equation? Why did you not have a separate, simultaneous equation for the water balance—that is, two equations, one for salt and one for water balance; why did you choose to combine them?

John Fryberger: I felt it was necessary to combine the two because we are considering a mass balance. The evapotranspiration loss would remove water from the alluvium and tend to concentrate the remaining water so it's necessary to include that in the over-all equation. There is a second equation that takes into consideration only the water and not the salt. The second equation is a simultaneous equation so you can solve for two unknowns but that's the only way it's used.

Maxine Goad: I still don't understand why you would have to have a term in there for the salt removed by evapotranspiration since it is salt we're balancing in the one equation.

John Fryberger: It's both salt and water in the main equation so I felt both factors had to be considered: the tons of salt and the amount of water because you have to have both factors in order to have concentrations. Now, the end result of the equation results in one factor being concentration at a given time interval after the salt inflow has been stopped so you're solving for concentration. The second thing you're after are tons of salt per year coming out of the alluvium that goes into the base flow. For the first factor, the concentration, I think it is necessary to include the evapotranspiration losses in the main equation in order to arrive at that concentration.

Elmer Jones: I think the two terms are necessary. If you are considering just the salt balance on a short-term basis I think you could say the precipitation contains a zero concentration; the evapotranspiration loss contains a zero concentration but over a period of years the precipitation falling on the outer edges of the Valley probably has provided a driving force to drive the saline water in towards your stream.

Maxine Goad: It's certainly true for concentration; concentration is related both to the amount of water and to the amount of salt.

Otto Helweg: So if this is true, you would have to include ET even though it affects concentration, but, of course, it doesn't affect the mass balance.

John Fryberger: Do I understand that you don't like the idea of having a value of one assigned to the evapotranspiration losses when really and truly the value ought to be zero?

Maxine Goad: Right.

John Fryberger: I agree with you; I don't like to assign a value of one when you know it's probably not one and that it is probably closer to zero, but I had to use one because otherwise the factor would drop out-- the E factor would drop out.

Tyler Gass: We have to adjourn now. Thank you for your participation.

DISCUSSION

The following questions were answered by W. B. Wilkinson after delivering his talk entitled "Prediction of Future Nitrate Concentrations in Ground Water." (The paper appears on pages 70-82.)

Q. by P. K. Saint. *Where NO_3-N levels exceed drinking water standards in potable aquifers, is the information being passed to health authorities, and what measures are being taken?*
A. Where the water authorities in England and Wales record nitrate levels in public supplies in excess of the recommended WHO's European drinking water standard of 50 mg/l (11.3 mg/l NO_3-N), the health authorities are informed. They, in turn, alert their staff, family practitioners and pediatricians within the area. Environmental health departments will also consider private supplies. The health authorities may also advise mothers who are bottle feeding young infants against excessive boiling of water for the infants' needs. If the health authority considers it necessary, supplies of low nitrate bottled water will be made available to infants. This provision is made in all cases where the nitrate levels exceed the 100 mg/l (22.6 mg/l NO_3-N), this being the WHO acceptable European Standards for drinking waters.

Health authorities are asked to immediately notify the Department of the Environment of any confirmed cases of methaemoglobinaemia so as to help the Department and the water authorities in planning future policy.

Q. *How were samples collected to prevent contamination from overlying portions of the aquifer?*
A. Preliminary samples from Chalk sites, and the majority of samples from Bunter Sandstone sites were obtained by power auger. Each sample was composed of cuttings from the auger flights obtained during penetration of a 0.5-m or 1-m (1½- or 3-ft) interval. The augers were withdrawn from the borehole between each sampling run and cleansed thoroughly. Temporary lining tubes were inserted in boreholes when unstable ground conditions threatened collapse of material at high levels on to the auger flights. Comparison of analyses on interstitial water from auger samples with analyses of water from adjacent solid cored holes suggested that contamination was minimal. Core samples were obtained from the Chalk by driving a 450-mm (18-in) long, 100-mm (4-in) diameter core barrel into the rock. After withdrawal of the core barrel, the borehole was enlarged to a diameter of 150 mm (6 in) to the base of the cored hole, drilling rubble cleared out and another core taken. Deep boreholes in the Bunter Sandstone were drilled by rotary air flush methods to produce a 100-mm (4-in) diameter core. Experience in other research programmes had shown that this method limited the invasion of the core by supernatant borehole liquids to the outer 1 to 2 cm (½ to 1 in) of Bunter Sandstone cores. Cores were extruded from the sample tubes on site. 5 cm (2 in) from each end was discarded and the remainder wrapped and sealed in two layers of polyethylene for transport to the laboratory. Auger samples were similarly packed for transport.

Q. by Don Runnells. *How was the water extracted from the unsaturated materials?* **Q. by C. Roberts.** *Could the high speed centrifugation of the sample possibly produce erroneously high (or low) values of nitrate and tritium because of the non-"in situ" or unnatural conditions of extraction?*
A. Approximately 300 grams of material were removed from the central parts of cores in the laboratory, excluding material from the outer 2.5 cm (1 in) of the core. The material was spun at 6,000 rpm in a refrigerated centrifuge at a temperature of 12°C for one hour. Samples of auger cuttings were processed in a similar manner. Chalk samples generally yielded between 10 and 25 ml of interstitial water and sandstone samples up to 10 ml. Replicate Chalk samples have been extracted by centrifugation at 4,500 and 6,000 rpm and by leaching disaggregated material in deionized water. Nitrate contents of the extracted waters were found to be comparable within the limits of experimental error. Electron micrographs of replicate Chalk samples taken before and after centrifugation have been examined for evidence of matrix distortion or pressure solution of the carbonate minerals, but none has been noted. It is concluded that the method of extraction does not affect the value of the nitrate concentration. Samples for tritium analysis were extracted by a standard vacuum distillation procedure at the Harwell Laboratory of the United Kingdom Atomic Energy Authority.

Q. by L. A. Swain. *To what do you attribute the high nitrate (105 mg/l NO_3-N) at shallow depth in the Chalk on the unfertilized grassland sites?*
A. The high nitrate concentrations measured at shallow depths in certain samples taken from unfertilized sites are attributed to partial mineralization of the organic nitrogen in the sample during the time between field extraction and laboratory analysis of the sample. Further investigations of this phenomenon, which appears to be limited to the upper 1 to 2 metres (3 to 6 ft), at Chalk sites, are in hand.

Q. by M. Johnson. *Is the tritium accumulation at 10 m a moving front or an accumulation area? If it is an accumulation area, why?*
A. No direct evidence is yet available for movement of the peak tritium accumulation in the Chalk in the United Kingdom, but circumstantial evidence suggests that the front is mobile. Smith *et al.* (1970) reported peak tritium

concentrations at a depth of 4 metres (13 ft) in the Chalk, and suggested a correlation with the 1963/64 thermonuclear tritium maxima. Tritium profiles measured on samples from boreholes drilled in the Chalk during 1974/75 exhibit peak concentrations at depths between 8 and 13 metres (26 and 43 ft) possibly indicating downward migration rates of between 0.80 and 1.20 metres (2.6 and 3.9 ft) per year. Sukhija and Shah (1976) have demonstrated the vertical movement of tritium fronts in alluvial sediments in the Indian subcontinent.

Q. by J. M. Klein. *Was any attempt made to determine if there was a varying nitrogen isotope ratio (N^{14}/N^{15}) in the pore waters collected from the varying land types and land uses?*
A. Measurement of nitrogen isotope ratios in pore waters has not yet been undertaken but are being planned.

Q. *How would you compare the nitrate in the Chalk with the same in sand and gravel?*
A. Studies have been concentrated predominantly on the Chalk aquifer. No work has been undertaken in unconsolidated sands or gravels. Sampling in the consolidated Bunter Sandstone indicates that nitrate concentrations in interstitial water are related to land usage in a manner similar to that for the Chalk, but detailed interpretation of the results have not yet been completed.

Q. *How did the actual distribution of nitrates in the aquifer correlate with the (1) land use pattern and (2) calculated values (from the model)?*
A. Detailed determination of the spatial distribution of nitrate in the unsaturated and saturated zones of an aquifer unit, and its relationship to land usage is the subject of current work. The simplified catchment model reported in the paper (Figure 20) shows good agreement between calculated and measured concentrations in the ground water and the actual land usage.

Q. by A. A. Baker. *What is the effect of dispersion on the predicted nitrate concentration?*
A. Dispersion was included in both the vertical flow model and in the catchment model. Its effect is to attenuate the peak concentrations. The most significant contribution to peak attenuation results from the variability of downward migration rates through the unsaturated zone, which may be considered as a macro-scale dispersion.

Q. by M. Apgar. *What parameters are required for computations in your model for prediction of future nitrate concentrations?*
A. The vertical flow model required annual infiltration and fertilizer application rates; land use records and the depth of unsaturated chalk. Because of the slow rate of solute movement through the unsaturated chalk, nitrate input to the water table in any year may only be predicted when data are available for earlier years. To run the catchment model these data must be specified for each zone of land use within the catchment. In addition the ground-water level distribution, the aquifer properties, and boundary conditions of the catchment are required.

Q. by P. O. Saman. *Did you also consider point-screens in your study, like sanitary landfills, etc.?*
A. Point sources were not considered in the investigations. Their contribution to nitrate contamination of ground water in the region studied is insignificant. The catchment model assumed complete mixing of water and nitrate in each zone which is acceptable for a distributed source, but not for point sources.

Q. by L. Rowe. *Did your prediction model allow for the resaturation of the aquifer immediately above the saturated zone?*
A. No; the model calculations were based on yearly inputs of rainfall and nitrate with the model years running from April to March inclusive. It was assumed that there would not be a significant change in water level from one year to the next.

Q. by H. Bouwer. *Would it be possible to reduce nitrate input by better fertiliser use?*
A. Fried *et al.* (1976) have recently summarized data showing that excessive fertilizer applications may lead to inefficient uptake by crops, leaving increasing residual nitrate liable to leaching. Greene and Walker (1970) have suggested that, under certain circumstances, restricting the rate of fertilizer applications may be accompanied by a reduction of nitrate concentration in Chalk ground water. Efficient management of fertilizers may reduce the excess nitrate available for leaching, and could include regulation of timing and quantities of applications to take account of individual crop requirements and residual amounts of nitrate remaining in the rooting zone from previous crops (Burns, 1975). However, Young, Hall and Oakes (1972) have postulated that the increases in nitrate content of some ground waters in the United Kingdom may be related to an increase in arable acreage during the period 1939-45, as a result of which organic nitrogen stored in the soil profiles of long-term pasture became available for mineralization and leaching. It is considered possible that changes from a predominantly pastoral to an arable economy on other areas may lead to similar increases in the rate of nitrate leaching.

Q. *Did you investigate any other chemical or biological reactions that might interfere with the nitrogen movement and form?* **Q. by M. Johnson.** *Have you ever detected denitrification taking place at the saturated zone?*
A. Investigation of potential biochemical nitrogen transformations in the unsaturated and saturated zones is being carried out and some of the preliminary results are described by Young *et al.*, 1976. No evidence has been observed, to date, of in situ denitrification in the saturated zone.

References

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